SOIL CARBON AND NITROGEN STOCKS

IN A SEMIARID URBAN ECOSYSTEM

by

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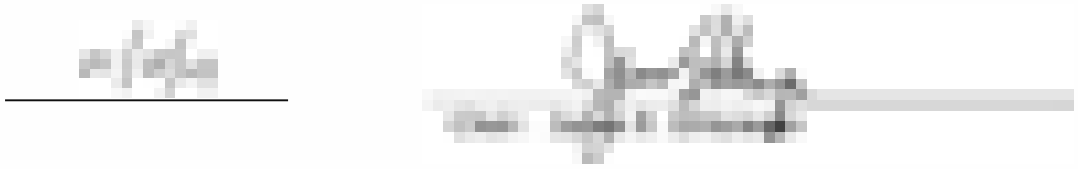

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ABSTRACT

The Earth's soils hold an estimated 1500 Pg of carbon and 100 Pg of nitrogen. Soils thus figure prominently into global C and N cycling. Human-related land cover changes have the potential to alter soil C and N pools, with implications for atmospheric CO_2 concentrations, ecosystem nitrogen availability, and other global and regional characteristics. Land cover modifications associated with urbanization are occurring at high rates in the USA and elsewhere, yet studies of such changes are currently underrepresented in the ecological literature. This thesis examines the effects of urbanization on soil C and N stocks in a semiarid region of the US. Soil samples were collected and analyzed from 38 residential parcels ranging in age from 7 to 100 years. Along this urban age gradient in the Salt Lake Valley, UT metropolitan area, soil organic C (SOC) and total N (TN) stocks increased significantly as a function of yard age. The strongest trends and largest gains were observed in the upper 20 cm beneath lawns. At this depth yard age was the site characteristic exerting the greatest influence on lawn SOC and TN stocks, eclipsing other factors such soil texture, prior land use and elevation. SOC and TN were highly correlated at all depths and beneath all pervious ground covers, indicating a close association between C and N cycling in this urban ecosystem. Inorganic C content of soils was highly variable both within and between sites and showed no trend with regard to yard age. For the upper 20 cm of residential lawns, the results presented here suggest a more than three-fold increase in average SOC and

more than two-fold increase in average TN following a century of urban land use. These gains exceed the average amount of C and N measured in aboveground tree biomass on residential study sites.

To my wife,
without whom this work would not have been done

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CHAPTER 1

INTRODUCTION

Urbanized landscapes are arguably some of the most managed ecosystems on earth. The intensity of management is such that even the placement of individual trees and shrubs is often governed by human design. Despite ponderous human influence, natural processes in these environments continue to function. Leafy ribbons of street trees exchange large amounts of atmospheric gases on a daily basis (Nowak 1994). Soils beneath manicured lawns provide habitat to countless fungi, bacteria, invertebrates and other small animals (Falk 1976; Kaye et al. 2005). Even in vacant lots, water and nutrients continue to cycle between the soil, plants, animals and atmosphere.

As ecosystems, modern cities are novel. Their influence on the composition of the atmosphere is highly disproportionate to their size. Dense human populations, coupled with high per capita rates of fossil fuel consumption, result in North American cities exporting enormous quantities of carbon dioxide and other greenhouse gases. George et al. (2007) reported an average atmospheric CO_2 difference of 66 ppm between adjacent urban and rural areas in the eastern US, illustrating the importance of urban systems with regard to global carbon cycling.

The novelty of urban ecosystems is not limited to their dense human populations. Often, vegetation in cities is markedly different from that found in surrounding environs. In the US and elsewhere, the livability of towns and cities has long been associated with

trees, lush lawns and tidy landscaping. Prior to widespread use of air conditioning beginning in the 1960s, traditional urban landscaping offered a means of cooling homes through shade and evapotranspiration, and urban vegetation continues to function in this capacity (Huang et al. 1992; Jo and McPherson 2001). It could be further argued that the westward trend of immigration in the US – from Europe to the eastern edge of the continent, and from the eastern US westward across the Great Plains – instilled in immigrants a proclivity for the deciduous trees and lawns of their home regions. Author and historian Wallace Stegner (1992) noted that it has taken Euro-American culture more than a hundred years to clearly portray western landscapes in art and writing (in other words, to see the West as Westerners), much less develop human habitations that take into account the unique aesthetics and hydrological limitations of the West.

In arid and semiarid portions of the western US, the results of homogenized landscaping preferences on regional vegetation are especially striking. Urban forests consisting largely of tree species from the eastern US and other continents have replaced open grassland, desert and shrub steppe. Managed turfgrass now covers an estimated 10,948 km^2 – or 39% of total urban land cover – in Arizona, New Mexico, Nevada, Utah, Colorado, Wyoming, Idaho and Montana combined (Nowak and Crane 2002; Milesi et al. 2005). Irrigation and nutrient inputs have been key factors facilitating this transformation, allowing urban residents to plant species beyond their natural ranges and to effectively extend the growing season in the urban environment. In many urban ecosystems, societal landscaping preferences seem to exert greater influence on vegetation characteristics than climate, soil properties, seed dispersal, competitive interactions, and other natural processes (Golubiewski 2006).

This thesis explores the impacts of urban land use conversions in the western US on soil biogeochemical cycling. This topic is on the one hand historical. How have past human activities influenced current conditions in the region? Throughout the Amazon basin, scientists have described a soil called Terra Preta (Amazonian Dark Earths) that on average contains three times more organic matter than surrounding soils (Glaser 2007; Solomon et al. 2007). The anomalous fertility of this soil is thought to have arisen from the intentional or unintentional practices of pre-Columbian native populations. Has an analogous legacy occurred in modern cities of the arid and semiarid US, where irrigated and fertilized landscapes have replaced native vegetation? Has the heightened productivity of urban vegetation resulted in concurrent changes in soil organic matter and other properties? A growing collection of studies suggests that such changes are likely (Qian and Follett 2002; Golubieski 2006; Lewis et al. 2006).

The research presented here is on the other hand prognostic. How will past and current activities influence conditions in the future? Are the ecological impacts of urbanization primarily local or do they have global implications? While management of urban landscapes can be considerable, seemingly little of this management is undertaken with the goal of maximizing ecosystem services and minimizing costs to human health and the environment (Byrne et al. 2008). Ecology and similar scientific disciplines have historically played little part in urban design. Indeed, the most populated landscapes on earth remain some of the least studied and most poorly understood. Increased research into the ecological consequences of various urban landscaping practices may help public workers and concerned citizens make more informed choices regarding urban development and conservation of natural resources.

Urbanization is a dominant land cover transformation occurring throughout much of the US. The proportion of the US population living in urban areas has grown over the past 50 years and is expected to increase (Pataki et al. 2006). Urban land cover has expanded at even faster rates than population, indicating a decreasing trend in housing density associated with ex-urban development (Hobbs and Stoops 2002). Moreover, recent population growth has not been evenly distributed across the US. Many of the fastest growing regions are also the most arid and nutrient poor. In the 1990s, for example, population growth in the West was 19.7%, compared to 5.5% in the Northeast and 7.9% in the Midwest (Perry and Mackun 2001). In fact, the western US experienced greater population growth in the 20th century than any other region, with the majority of this growth occurring in metropolitan areas (Hobbs and Stoops 2002).

Such trends suggest that human impacts to natural resources and ecological processes will be increasingly tied to urban development (Grimm et al. 2000; Pataki et al. 2006; Hall et al. 2008; Raciti et al. 2008). The consequences of urbanization may manifest as both the "ecological footprints" of urban residents, as well as direct conversion of native and agricultural ecosystems to urban land cover (Lambin et al. 2001). Despite the fact that more than 80% of the US population lives in metropolitan areas as defined by the US Census Bureau (Perry and Mackun 2001), the impact of urban land cover transformations on even basic ecosystem properties, such as the nutrient content of soils, remains only poorly understood. Even less is known about how these direct impacts might vary by region and climate.

This thesis attempts to enhance current knowledge about ecosystem responses to urban land cover change, with a focus on urban development in semiarid climates. Here, I

report on efforts to quantify changes in soil carbon and nitrogen stocks following urbanization of the semiarid Salt Lake Valley, UT. The Salt Lake Valley metropolitan area and its satellite cities to the north and south comprise the largest urban area in the Great Basin region. More than 80% of Utah's population currently resides in a chain of valleys flanking the western edge of the Wasatch Mountains (Fig. 1.1, 1.2). Similar to other urban areas across the western US, the Salt Lake Valley has undergone extensive conversion from grassland, shrubland and farmland to a matrix of impervious surfaces and urban forests characterized by abundant tree cover and lawn. At the foundation of this research was the hypothesis that soil C stocks should increase over time beneath pervious surfaces of the Salt Lake Valley urban ecosystem. This hypothesis arose from both theoretical and empirical evidence.

First, plant productivity and biomass in semiarid climates is typically limited by water availability (Neary et al. 2003). Organic C content of soils (that is, C derived from plants or animals) is most fundamentally controlled by the relative rates of litter input and litter decomposition. Litter inputs bring atmospheric C fixed by plants into the soil system, while microorganisms respire CO_2 back to the atmosphere during litter decomposition. The typically low plant productivity of arid and semiarid regions limits the amount of organic material entering soils and contributes to geographic trends characterized by decreasing soil organic carbon (SOC) stocks with decreasing precipitation (Jobbágy and Jackson 2000; Homann 2007). The high productivity of turfgrass suggests that both litter inputs and SOC stocks should increase following conversion of semiarid vegetation to urban lawn (Falk 1976; Falk 1980; Kaye et al. 2005). While agricultural crops may exhibit comparable rates of primary productivity

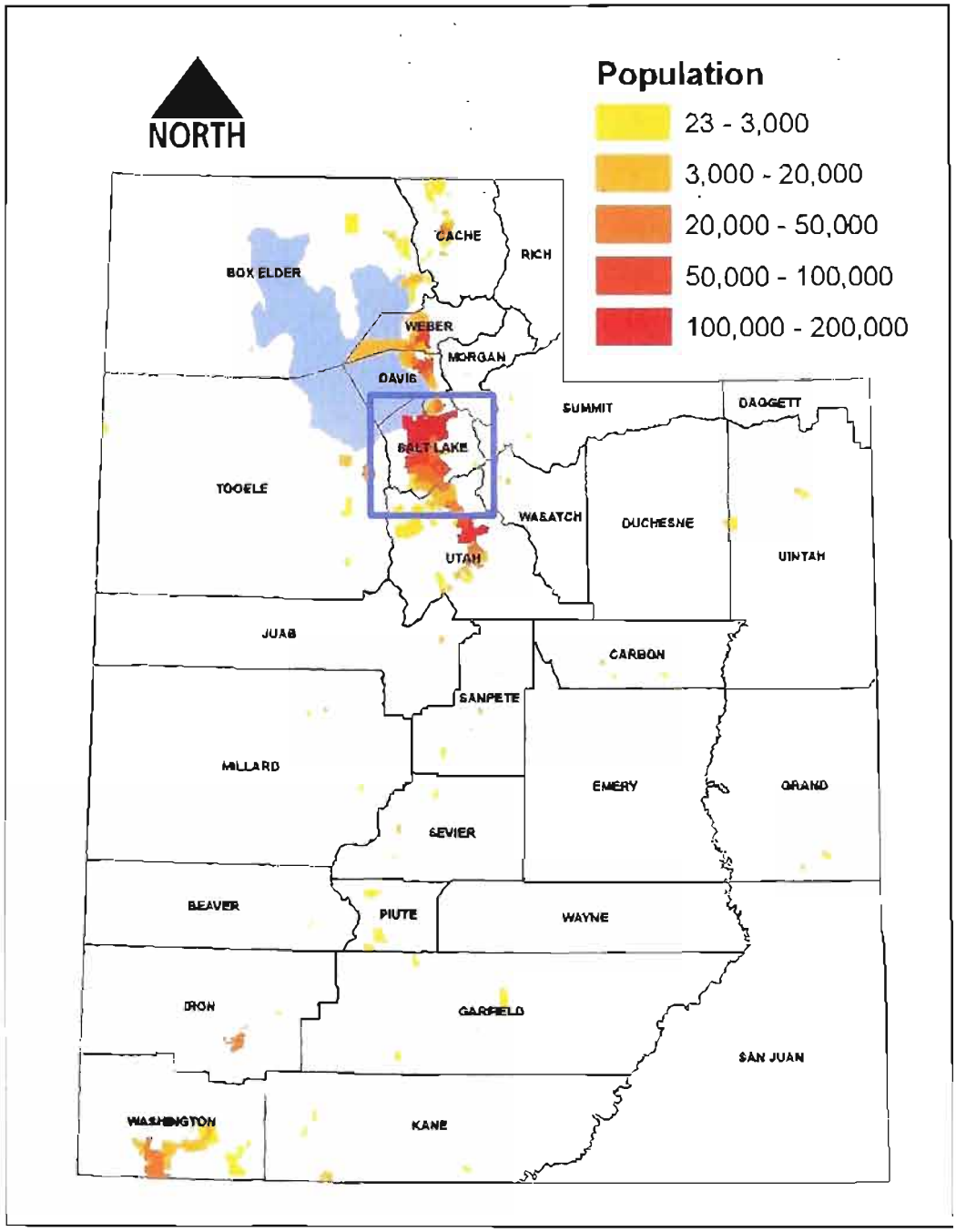

Fig. 1.1 Distribution of the Utah population reported by the 2000 census. The blue square highlights the study region.

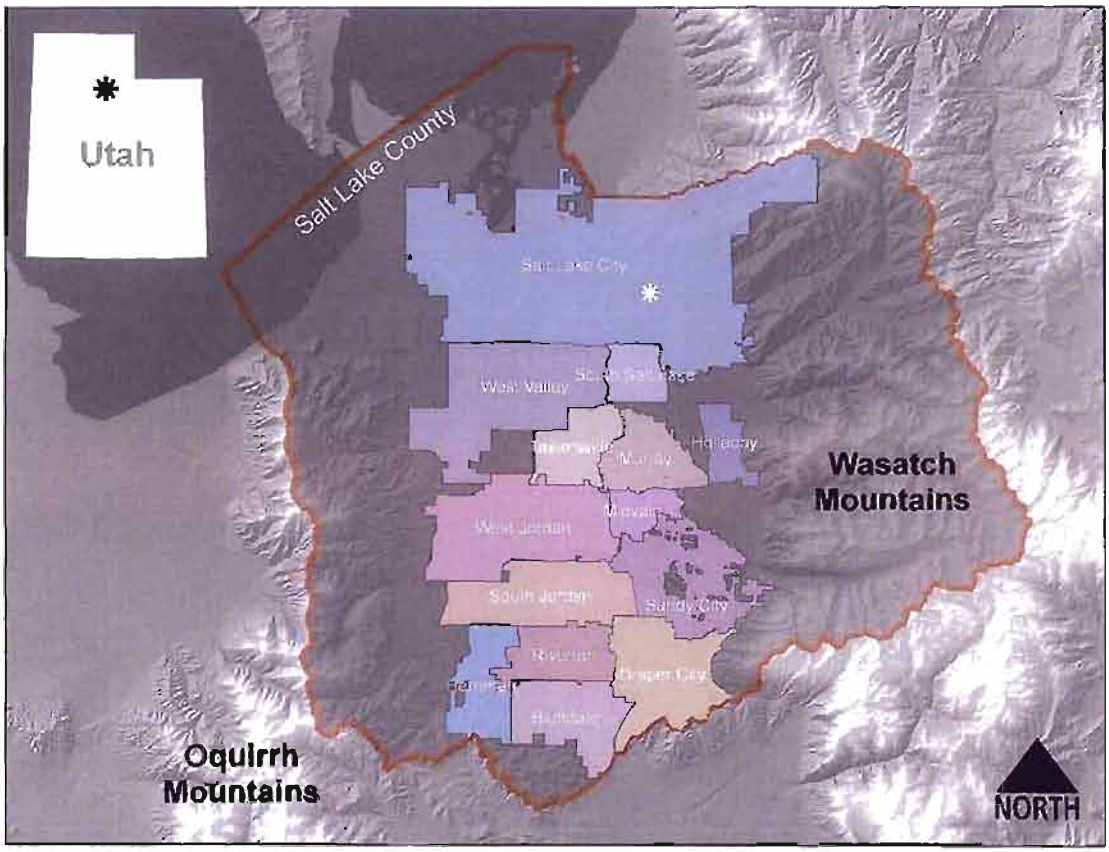

Fig. 1.2 Salt Lake Valley, USA metropolitan area, showing the boundaries of incorporated cities.

(Kaye et al. 2005), much of this biomass is harvested, and regular soil disturbance associated with tilling often results in high rates of C loss via decomposition (Davidson and Ackerman 1993). Lawns established on farmland typically do not experience this level of disturbance and may therefore be expected to accumulate SOC much like those replacing native vegetation. These theoretical expectations are supported by limited field measurements showing significant gains in SOC as lawns age. Such trends have been reported for residential yard in northern Colorado (Golubieswki 2006) and for golf courses in Colorado, Wyoming and North Carolina (Qian and Follett 2002; Wei Shi 2006).

Turfgrass and other mesic vegetation are often maintained in unfavorable climates by large inputs of water and fertilizer. The prevalent use of N amendments, together with the observation that most N in soil is associated with organic molecules (Post et al. 1985), suggests that accumulation of organic matter beneath lawns may facilitate accumulation of N as well. C and N are predicted to aggrade beneath pervious surfaces of urbanized landscapes for similar reasons. Yet the societal and ecological consequences of these changes may be considerably different. Sequestration of C in soil represents the removal of C from the atmosphere. While it is widely understood that fossil fuel use and deforestation have contributed to a greater than 30% increase in atmospheric carbon dioxide since the start of the Industrial Revolution, less publicized is the fact that 40-50% of anthropogenic carbon dioxide emissions never accumulate in the atmosphere (Solomon et al. 2007). Terrestrial vegetation and the earth's oceans appear to absorb the difference. Understanding the size, location and character of these carbon sinks is important, since it is not clear for how long and to what extent oceans and terrestrial

ecosystems will continue to offset anthropogenic carbon emissions (Schimel et al. 2001). Expansion of turfgrass in arid and semiarid regions has been suggested as a potential sink for atmospheric CO_2 (Qian and Follett 2002), but little information is currently available on the magnitude and duration of C sequestration by urban soils.

While the ability of lawns to sequester C may be desirable, the maintenance of these lawns with herbicides and fertilizers raises concerns about their impacts on water quality (Petrovik 1990). Nitrogen inputs to urban soils may be further supplemented by deposition of reactive N released to the atmosphere by fossil fuel combustion and agricultural activities. Balancing the difference between N inputs and outputs is the potential retention of N in vegetation and soils. Studies using ^{15}N as a tracer have found 9 to 51% of labeled fertilizer N in turfgrass soils several months to several years after application (Petrovic et al. 1990; Miltner et al. 1996; Engelsjord et al. 2004; Frank et al. 2006; Raciti et al. 2008). Studies examining the N content of urban soils of varying ages also suggest a high capacity of lawn soil to retain applied N (Porter et al. 1980; Lewis et al. 2006). Understanding the magnitude of soil N storage may be important for both minimizing environmental impacts of turfgrass fertilization and constraining the N balance of entire urban ecosystems (Baker et al. 2001).

In recognition of the different implications of C and N accrual in urban soils, this thesis explores each process in separate chapters. Chapter 1 presents and discusses findings related to SOC storage, while Chapter 2 addresses impacts of urbanization on soil N stocks. Each chapter is self-contained and written in the style of a refereed journal article. A third chapter is included to compare measurements of soil C and N storage at urban study sites to measurements of aboveground tree biomass. All three chapters report

on field data collected in the Salt Lake Valley urban ecosystem in 2007. Soil samples and tree measurements were collected from 38-40 residential parcels in the Salt Lake Valley ranging in age from 7 to 100 years old (Fig. 1.3). Trends observed along this urban age gradient were used to infer changes in C and N stocks following conversion of farmland and rangeland to residential use.

A traditional chronosequence is a series of study sites that that differ only in age, with all other factors important to soil formation (parent material, climate, topography, potential biota) being held constant (Stevens and Walker 1970). Although chronosequences are typically used to study soil development over periods greater than the human lifespan, large cities offer a unique opportunity to study changes in soil properties over shorter time scales, since altered soils spanning a range of ages can generally be found within a relatively small aerial extent. The phrase "urban age gradient" is used throughout this thesis in order to maintain the precise definition of a chronosequence and to account for the fact that the study reported here did not explicitly control for soil type and topographic position of sites. The reason for this deviation from a strict chronosequence was two-fold: 1) to gain a better understanding of the variation in soil properties throughout the Salt Lake Valley urban ecosystem, and 2) to gauge the importance of development age with regard to other common controls on soil carbon and nitrogen stocks. Soil texture, for example, has been shown to influence SOC stocks in many ecosystems (Nichols 1984; Burke et al. 1989; Jobbágy and Jackson 2000; Homann et al. 2007) and varies considerably within the Salt Lake Valley.

Chronosequence studies capitalize on a distinct feature of certain landscapes: the embodiment of ecosystem age as a function of space. Time and geography are two

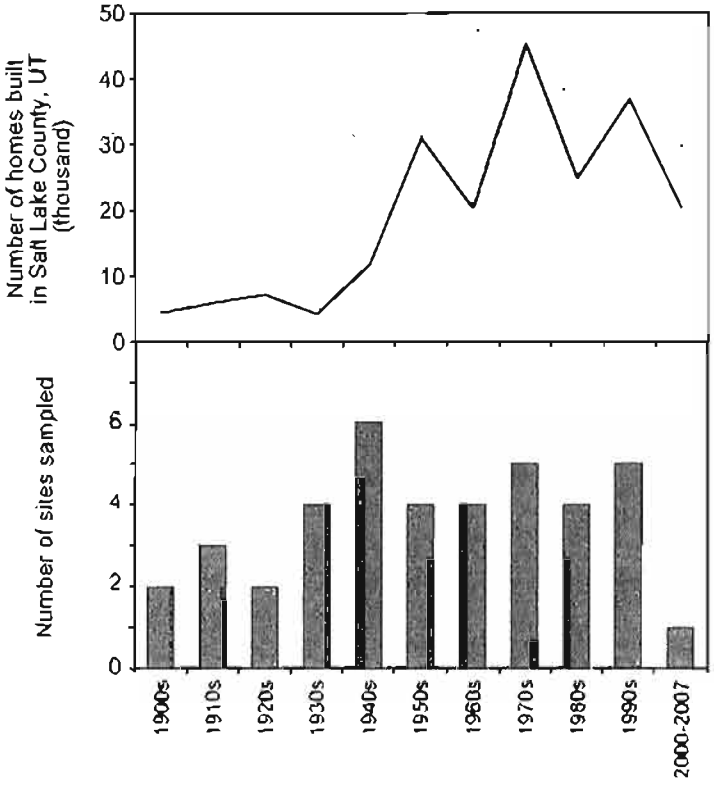

Fig. 1.3 Comparison of the age distribution of single-family homes in Salt Lake County, USA and the age distribution of homes sampled. Values shown are the number of homes built in each decade. Statistics on home construction were obtained from the Salt Lake County Assessor (2007).

themes diffusing much of the research presented here. On the one hand are questions regarding the legacies of past human activities and their implications for current and future ecosystem functioning. As the US population continues to grow and become more metropolitan, the sustainability of traditional urban landscape design may come into question, particularly with regard to outdoor water use and the application of herbicides and fertilizers. What can we learn about the ecological services and disservices of historic landscaping practices, and how can society use this knowledge to direct future urban design?

On the other hand are questions regarding geography and scale. Even if urbanization has led to considerable changes in soil biogeochemistry at a local level, what impact do these changes have on global processes such as the accumulation of CO_2 in the atmosphere? If C and N aggrade in urban soil over time, how do historic development patterns influence the spatial distribution of soil C and N stocks across an urbanized landscape? Perhaps even more basic, in what ways does climate influence the biological consequences of urbanization? Questions such as these provide a backdrop to the more rigorous hypotheses tackled in the following chapters.

References

Baker LA, Hope D, Xu Y, Edmonds J, Lauver L (2001) Nitrogen balance for the Central Arizona-Phoenix (CAP) ecosystem. Ecosystems 4:582-602

Burke, I.C., C.M. Yonker, W.J. Parton, C.V. Cole, K. Flach, and D.S. Schimel. 1989. Texture, climate, and cultivation effects on soil organic matter in U.S. grassland soils. Soil Science Society of America Journal 53:800-805

Byrne LB, Bruns MA, Kim KC (2008) Ecosystem properties of urban land covers at the aboveground-belowground interface. Ecosystems DOI: 10.1007/s10021-008-9179-3

Davidson EA, Ackerman IL (1993) Changes in soil carbon inventories following cultivation of previously untilled soils. Biogeochemistry 20:161-193

Engelsjord ME, Branham BE, Horgan BP (2004) The fate of nitrogen-15 ammonium sulfate applied to Kentucky bluegrass and perennial ryegrass turfs. Crop Science 44:1341-1347

Falk JH (1976) Energetics of a suburban lawn ecosystem. Ecology 57:141-150

Falk JH (1980) The primary productivity of lawns in a temperate environment. Journal of Applied Ecology 17:689-695

Frank KW, O'Reilly KM, Crum JR, Calhoun RN (2006) The fate of nitrogen applied to a mature Kentucky bluegrass turf. Crop Science 46:209-215

George K, Ziska LH, Bunce JA, Quebedeaux B (2007) Elevated atmospheric CO_2 concentration and temperature across an urban-rural transect. Atmospheric Environment 41:7654-7665

Glaser B (2007) Prehistorically modified soils of central Amazonia: a model for sustainable agriculture in the twenty-first century. Philosophical transactions of the Royal Society 362:187-196

Golubiewski NE (2006) Urbanization transforms prairie carbon pools: effects of landscaping in Colorado's Front Range. Ecological Applications 16:555-571

Gimm NB, Grove JM, Pickett STA, Redman CL (2000) Integrated approaches to long-term studies of urban ecological systems. BioSceince 50:571-584

Hall SJ, Huber D, Grimm NB (2008) Soil N_2O and NO emissions from an arid, urban ecosystem. Journal of Geophysical Research 113: G01016, doi:10.1029/2007JG000523

Hobbs F and N Stoops (2002) U.S. Census Bureau, Census 2000 Special Reports, Series CENSR-4, Demographic Trends in the 20th Century, U.S. Government Printing Office, Washington, DC, 2002

Homann PS, Kapchinske JS, Boyce A (2007) Relations of mineral-soil C and N to climate and texture: regional differences within the conterminous USA. Biogeochemistry 85:303-316

Huang J, Ritschard R, Sampson N, Taha H (1992) The benefits of urban trees. In: Akbari H, Davis S, Dorsano S, Huang J, Winnett S (eds) Cooling our communities. US Environmental Protection Agency, Washington, DC, pp 27-42.

Jo H-K, McPherson EG (2001) Indirect carbon reduction by residential vegetation and planting strategies in Chicago, USA. Journal of Environmental Management (2001) 61:165-177

Jobággy EG and Jackson RB (2000) The vertical distribution of soil organic carbon and its relation to climate and vegetation. Ecological Applications 10:423-436

Kaye JP, McCulley RL, Burke IC (2005) Carbon fluxes, nitrogen cycling, and soil microbial communities in adjacent urban, native and agricultural ecosystems. Global Change Biology 11:575-587

Lambin EF et al. (2001) The causes of land-use and land-cover change: Moving beyond the myths. Global Environmental Change 11:261-269

Lewis, D.B., J.P. Kaye, C. Gries, A.P. Kinzig, and C.L. Redman. 2006. Agrarian legacy in soil nutrient pools of urbanizing arid lands. Global Change Biology 12:703-709

Milesi C, Running SW, Elvidge CD, Dietz JB, Tuttle BT, Nemani RR (2005) Mapping and modeling the biogeochemical cycling of turf grasses in the United States. Environ Manage 36:426-438

Miltner ED, Branham BE, Paul EA, Rieke PE (1996) Leaching and mass balance of ^{15}N-labeled urea applied to a Kentucky bluegrass turf. Crop Science 36:1427-1433

Neary DG, Overby ST, Hart SC (2003) Soil carbon in arid and semiarid forest ecosystems. In Kimble JM, Heath LS, Birdsey RA, Lal R (eds) The potential for U.S. forest soils to sequester carbon and mitigate the greenhouse effect. Lewis Publishers, Boca Raton, FL, pp 293-308

Nichols JD (1984) Relation of organic carbon to soil properties and climate in the southern Great Plains. Soil Science Society of America Journal 48:13821384

Nowak DJ (1994) Air pollution removal by Chicago's urban forest. In: McPherson EG, Nowak DJ, Rowntree, RA (eds) Chicago's urban forest ecosystem: results of the Chicago Urban Forest Climate Project. USDA Forest Service General Technical Report NE-186

Nowak DJ, Crane DE (2002) Carbon storage and sequestration by urban trees in the USA. Environmental Pollution 116:381-389

Perry MJ and PJ Mackun (2001) U.S. Census Bureau, Census 2000 Brief, Series C2KBR/01-2, Population Change and Distribution, U.S. Government Printing Office, Washington, DC, 2002.

Pataki DE, Alig RJ, Fung AS, Golubiewski NE, Kennedy CA, McPherson EG, Nowak DJ, Pouyat RV, Romero Lankao P (2006) Urban ecosystems and the North

American carbon cycle. Global Change Biology 12:2092-2102

Petrovic AM (1990) The fate of nitrogenous fertilizers applied to turfgrass. Journal of Environmental Quality 19:1-14

Porter KS, Bouldin DR, Pacenka S, Kossack RS, Shoemaker CA, Pucci AA Jr (1980) Studies to assess the fate of nitrogen applied to turf: Part 1. Research Project Technical Completion Report OWRT Project A-086-NY. Cornell University

Post WM, Pastor J, Zinke PJ, Stangenberger AG (1985) Global patterns of soil nitrogen storage. Nature 317:613-616

Qian Y, Follett RF (2002) Assessing soil carbon sequestration in turfgrass systems using long-term soil testing data. Agronomy Journal 94:930-935

Raciti SM, Groffman PM, Fahey TJ (2008) Nitrogen retention in urban lawns and forest. Ecological Applications 18:1615-1626

Schimel et al. 2001. Recent patterns and mechanisms of carbon exchange by terrestrial ecosystems. Nature 414:169-172

Shi W, Dell E, Bowman D, Iyyemperumal K (2006) Soil enzyme activities and organic matter composition in a turfgrass chronosequence. Plant Soil 288:285-296

Solomon D, Lehmann J, Thies J, Schäfer T, Liang B, Kinyangi J, Neves E, Petersen J, Luizão F, Skjemstad J (2007) Molecular signature and sources of biochemical recalcitrance of organic C in Amazonian Dark Earths. Geochimica et Cosmochimica Acta 71:2285-2298

Solomon S et al. (2007) Technical summary. In Solomon S, Qin D, Manning M, Chen Z, Marquis M, Averyt KB, Tignor M, Miller HL (eds) Climate change 2007: The physical science basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, New York, NY

Stegner W (1992) Where the bluebird sings to lemonade springs. Random House, Inc., New York, NY

Stevens PR, Walker TW (1970) The chronosequence concept and soil formation. The Quarterly Review of Biology 45:333-350

CHAPTER 2

SOIL CARBON STOCKS ALONG AN URBAN AGE GRADIENT IN SEMIARID SALT LAKE VALLEY, UT

Abstract

In many cities of the arid and semiarid USA, societal preferences for shade trees and lawns have led to dramatic shifts in regional vegetation. Heightened productivity of urban vegetation may promote accumulation of organic carbon in soils beneath lawn and other pervious ground cover. I measured soil carbon stocks along an urban development age gradient in the Salt Lake Valley, USA metropolitan area. Soils were sampled from 38 single-family residential parcels ranging in age from 7 to 100 years. A highly significant positive relationship was observed between time since development and organic carbon content (SOC) of surface soils (0-20 cm) beneath residential lawns ($r^2 = 0.64$). Predicted SOC to this depth was more than three-fold greater at the oldest sites than the youngest sites based on standard linear regression. My results suggest that SOC stocks may increase beneath turfgrass in semiarid regions for longer than previously reported. Yard age explained considerably more variability in lawn SOC to 20 cm than did soil texture, elevation or other site characteristics, but the importance of this variable diminished with depth. SOC beneath residential pervious surfaces other than turf also rose significantly as a function of yard age. High variability in nonturf SOC among sites reduced the strength of this relationship, however. Empirical modeling suggests that future regional gains in

SOC stocks beneath residential lawns in the Salt Lake Valley will far exceed gains achieved thus far, due to prolific urban growth in the latter half of the 20th century.

Introduction

Water availability exerts a fundamental control on many ecosystem processes (Austin and Vitousek 1998). In arid and semiarid regions of the US, urban development often alters ecosystem water availability though widespread use of irrigation. Societal preferences for shade trees and lawns, made possible through supplemental water and nutrient inputs, have resulted in extensive modification of regional vegetation in many areas. Managed turfgrass now covers an estimated 10,948 km^2 in Arizona, Colorado, Idaho, Montana, Nevada, New Mexico, Utah and Wyoming combined (Milesi et al. 2005). As much as 70% of annual household water use may go towards maintaining turfgrass and other horticultural species (Wall 2001). Such irrigation requirements generally far exceed the natural precipitation of most arid and semiarid regions, resulting in conversion of many urbanized landscapes from arid to humid environments. Climate is a major driver of plant productivity and soil C and N storage on global and regional scales (Jenny 1941; Post et al. 1985; Schlesinger 1991; Homann et al. 2007). Profound changes in water availability such as those found in urban landscapes of the western US may have important effects on soil organic and inorganic C storage, especially when accompanied by high levels of supplemental nitrogen.

About 40% of global soil organic carbon (SOC) resides in the upper 40 cm of the soil profile (Jobbágy and Jackson 2000), where land use change often has its greatest impact. SOC losses following conversion of temperate grassland and forest to agriculture are well documented (Aguilar et al. 1988; Burke 1989; Davidson and Ackerman 1993).

In arid regions, irrigated agriculture may have the opposite effect, augmenting SOC pools by increasing plant detritus inputs to otherwise low productivity systems (Mann 1986; Post and Mann 1990). Given that managed turfgrass now occupies more area of the US than any other irrigated crop (Milesi et al. 2005), the ecological importance of urban land use conversion may not be fully recognized. Like agriculture, urban development often physically disturbs the soil and leads to altered water and nutrient inputs to urbanized and adjacent ecosystems. Despite high rates of urban expansion in the US and elsewhere, the impact of this land use change on soil C stocks has been much less studied than for agricultural systems.

Rates of organic matter input and decomposition are thought to ultimately govern the size of SOC stocks at steady state. In turn, these processes respond asymmetrically to geographic variation in precipitation and temperature. Across global and regional climate gradients, SOC generally increases with mean annual precipitation and decreases with mean annual temperature (Jobbágy and Jackson 2000; Homann et al. 2002). SOC storage tends to be low in arid ecosystems due in part to water limitations on plant productivity and hence, plant inputs to the soil. In urban areas, irrigation and nutrient inputs may obscure regional variation in precipitation and N cycling. Indeed, the general homogeneity of turfgrass systems suggests that net primary productivity of this vegetation should have little relationship to natural precipitation, and that SOC storage beneath lawns in arid and semiarid regions may increase due to the heightened productivity of lawns compared to natural vegetation (Pouyat et al. 2003). Net gains in lawn SOC are likely to be a function of SOC inventories prior to urban development, as well as site-specific soil, climatic, topographic and anthropogenic variables that influence

organic matter input and decomposition and thus the size of SOC stocks at steady state. All SOC accrued through this process would involve removal of C from the atmosphere, suggesting that land cover changes related to urbanization may constitute an important component of regional C accounting (Qian and Follett 2002; Kaye et al. 2005).

Accumulation of SOC beneath both golf courses and residential lawns has been documented in the US Great Plains. Qian and Follett (2002) analyzed historic soil testing records for golf courses in Colorado and Wyoming. They found carbon sequestration to be between 0.9 and 1.0 t ha^{-1} yr^{-1}during the first 25 to 30 years after turfgrass establishment. Sampling along a chronosequence of residential urban development in the Denver metropolitan area, Golubiewski (2006) reported a greater than two-fold increase in average lawn soil organic carbon (0-30 cm) between homes built in the 1990s and homes built in the 1950s. A comparison of net primary production, C storage, and soil respiration among mature lawn, native grassland, and farmland in northern Colorado also found SOC stocks to be highest beneath lawns, despite greatly enhanced soil respiration at the urban sites (Kaye et al. 2005).

These and related model-based studies (Bandaranayake 2003; Milesi et al. 2005) demonstrate the potential for turfgrass landscapes to accumulate considerable amounts of SOC in semiarid regions. The effects of urbanization on total soil C storage (i.e. organic and inorganic C) are less clear. Global patterns of soil inorganic carbon (SIC) storage are opposite those described above for SOC. Less than 1% of global SIC is thought to be located in humid or per-humid regions; most of the remainder occurs in arid and semiarid climates (Eswaran et al. 2000). If these geographic trends are extended to the temporal conversion of arid ecosystems to mesic urban landscaping, then urbanization might be

expected to impact regional SIC stores. This simple prediction is complicated by the various controls on carbonate formation and dissolution (Khokhlova et al. 1997), by the chemical properties of applied irrigation water (Schlesinger 2000), and by general uncertainties regarding the effects of irrigation on SIC dynamics (Eshel et al. 2007). Also unclear is the extent to which C sequestration by managed turfgrass can offset anthropogenic C emissions. Urban areas currently occupy less than 3.5% of the conterminous US, only a fraction of which is located in dry areas (Nowak and Crane 2002). In the absence of very large changes in C stocks, contribution of urban soils to the removal of atmospheric CO_2 is probably constrained by the limited extent of urban and suburban development.

This research was initiated to assess changes in soil C stocks following urbanization of a semiarid region. Replacement of farmland and native plant communities with managed urban vegetation, as is occurring rapidly throughout the western US, may increase SOC storage by enhancing litter inputs and limiting soil disturbance. I suspected that it might also affect SIC stocks through outdoor irrigation and subsequent modifications of soil water balance. I attempted to quantify such changes by comparing soil C pools along an urban development age gradient in the Salt Lake Valley, USA metropolitan area. I analyzed soil samples from 38 single-family residential parcels ranging in age from 7 to 100 years, as well 3 farmland sites, with the expectation that SOC storage beneath lawns and other productive urban vegetation would increase as a function of age. Regression analysis was performed to identify site characteristics explaining the greatest amount of variability in soil C storage. Trends identified in this

analysis were then used to model historic and future changes in SOC beneath residential lawn cover within the study area.

Materials and methods

Residential study sites

Salt Lake Valley, UT and its satellite cities to the north and south form the largest metropolitan area in the Great Basin region. More than 80% of Utah's population resides in a chain of valleys flanking the western edge of the Wasatch Mountains. Here, the abrupt rise of the mountains confines most urban development to a narrow elevation range within the valley bottoms. I established 38 sites near the center of this densely populated region, within the urbanized portions of Salt Lake County (Fig. 2.1). The study area has a semiarid climate characterized by mean annual precipitation of 397 mm and mean average temperature of 11 degrees C (WorldClimate 2008). Early writings describe the area as primarily grassland prior to European settlement in the 1840s, with trees generally confined to riparian corridors and upland shrubs being much less dominant than in modern times (Wakefield 1933). Agriculture and grazing were the primary drivers of land use change until about 1950, when farmland reached its greatest aerial extent and urban development began to proliferate (US Census Bureau 1952). Agricultural census data suggests that virtually the entire study area was at some point either tilled for crops or grazed, meaning that soil carbon stocks may have been extensively altered from pre-settlement conditions prior to urbanization. Urban growth peaked in the 1970s and afterward began a shallow decline (Salt Lake County Assessor 2007). Farmland now occupies about 5% of Salt Lake County, compared to about 30% in urban land cover and 21% in federal ownership.

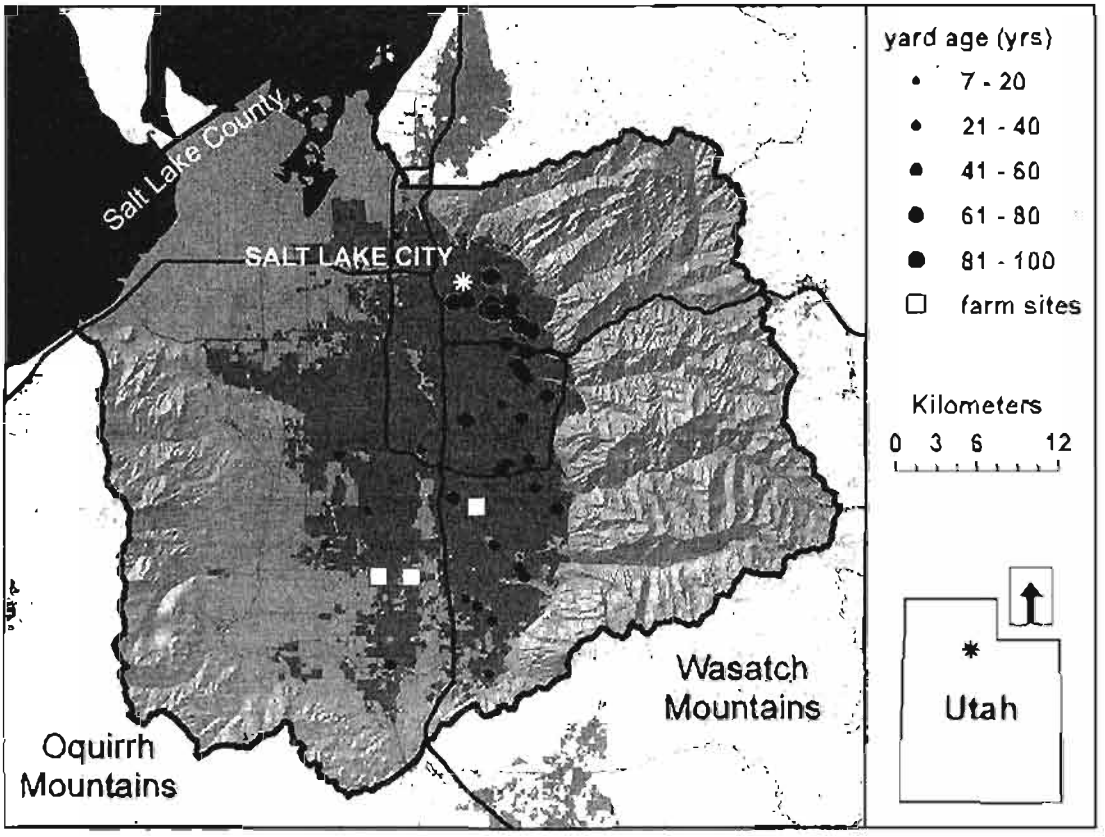

Fig. 2.1 Salt Lake Valley, USA and study site locations. Larger circles indicate older residential sites, based on the age of the current home. Urban land cover from the National Land Cover Dataset 2001 is shown in medium grey (Homer et al. 2007).

Residential sites were sampled between May and November of 2007. I sampled only single-family residential parcels, which in 2007 constituted approximately 39% of urban land cover within the study area. Potential study sites were randomly selected from a Salt Lake County parcel dataset using a geographic information system (GIS). Initial selection was stratified by decade of development, with the age of the current home used as a proxy for the date of transition to residential land use. Due to the time required to collect tree information on large parcels (data not reported here, but concurrently obtained for related research) parcels larger than 0.25 hectares were rejected. Random selection of parcels developed later than 1980 was also stratified by prior land use (irrigated agriculture, dryland agriculture, or range/wildland), as determined from historic aerial photography. Letters were sent to the owners of selected parcels soliciting permission to access their property. The number of parcels sampled in each decade was meant to reflect the current age distribution of single-family homes in Salt Lake County. Throughout the summer field campaign, properties were randomly selected and their owners contacted until a predefined quota was approached or met. Permission was granted for a total of 40 parcels representing all decades between 1900 and 2007. Soils could not be collected from one site due to time constraints, and another site had soils too shallow to adequately sample.

Study sites were dispersed throughout the Salt Lake Valley metropolitan area (Fig. 2.1). The maximum distance between sites was 33 km and the elevation range was 240 m. Aerial photography taken in 1958 was used to determine land use at sites prior to urban development. Of sites developed since 1959, 11 were constructed on farmland and 8 replaced undeveloped land or range. Because I controlled only for parcel age and prior

land use, sites varied with regard to parent material, soil texture, tree cover, yard size, lawn quality and other characteristics. Of 19 soil Great Groups found within the study area, my residential sites represented 8, although representation was unbalanced (Soil Survey Staff 2007). Forty percent of the soils in this area are classified as Mollisols, and 79% of my sites had parent soil within this soil order.

Soil sampling

At each residential parcel, three soil cores were collected to a depth of 40 cm using a 2 cm diameter soil recovery probe (AMS, Idaho Falls, ID). Two cores came from turfgrass lawn and the third from a nonturf area such as flowerbed, shrub row, vegetable garden, or natural vegetation. Lawn sampling points were randomly generated using a GIS prior to site visits, and they were located in the field using azimuth and distance calculated from obvious reference features. One core each was taken from front and rear lawns when possible. Front and rear yards were delineated in the GIS using digital parcel data and fence lines apparent on high resolution aerial photography. The boundary between front and rear yards was defined by fences, or by a line drawn through the center of the home when no fences were apparent. Five ranked points were generated for each yard, and these points were visited in order until a suitable sampling point (i.e. one located in turfgrass with no obvious conflicts with underground utilities) was found. Due to difficulty identifying nonturf areas from aerial photography, the sampling point for these areas was arbitrarily selected at the time of the field visit. The total number of cores collected for the study is listed in Table 2.1 by ground cover type. Deviations from the study design described above include one site where no nonturf core was collected, three sites where a nonturf core was substituted for a lawn core, and two sites where an

Table 2.1 Ground cover at soil sampling points

Ground cover	No. of cores
Residential turf	74
Front lawn	44
Rear lawn	30
Residential nonturf	35
Horticultural forbs/shrubs	19
Wild/native vegetation	7
Bare soil	3
Organic mulch/duff (<5% plant cover)	3
Vegetable garden	3
Agricultural	10
Irrigated grain	2
Irrigated hay	4
Pasture	4

additional lawn core was substituted for a nonturf core.

Cores were collected in 10 cm depth increments and removed from the sampling probe after each increment to reduce compaction. Rocky soils prevented extraction of one or more increment at the majority of sites. Soil samples containing at least 70% of a full increment were retained and the depth recorded. Otherwise, I returned partial increments to the excavation and made no further effort to collect a complete core. The percentage of samples successfully retrieved dropped considerably with depth (96% of 0-10 cm increments, 68% of 10-20 cm increments, 40% of 30-40 cm increments, and 33% of 30-40 cm increments). Others have noted the challenges of sampling soils in an urban setting (Jo and McPherson 1995; Golubiewski 2006).

Soil analysis

Each 10 cm core increment was individually processed and analyzed. Samples were air dried and passed through a 2 mm sieve to isolate the fine soil fraction. During sieving, increments were cleaned of visible roots using tweezers, and fine soil (<2 mm), coarse material (>2 mm), and fine root fractions (<2 mm diameter) were weighed. Two 5 g subsamples were isolated from air-dried and sieved soil through quartering (Tan 1996). One subsample was ground on a ball mill (MM200, Retch, Haan, Germany) for C analysis, and the other was dried in an 80 degree Celsius forced-air oven for 48 hours to derive a moisture correction factor for soil texture and bulk density calculations. Total C was measured on 30 (±1) mg of air-dried and ground soil using an elemental analyzer, inorganic C was measured on 300 (±10) mg of air-dried and ground soil via pressure calcimeter (Wagner et al. 1998, Sherrod et al. 2002), and organic C was determined by difference. SOC and SIC concentrations were converted to mass per unit ground surface area, integrated across each 10 cm depth, using the equation

$$\text{SOC (g m}^{-2}) = \%\ \text{SOC}/100 * F * (10\ \text{cm}/D)/A * 10\ 000 \qquad (2.1)$$

where F is the fine fraction weight (g), D is the increment depth recovered (cm), and A is the cross-sectional area of the sampling probe tip (cm^2). Because this calculation does not include stones larger than 2 cm, it likely overestimates soil C content in most cases. When comparisons were made among sites, C content of replicate cores was first averaged by depth, and then summed to report a single value for surface (0-20 cm) and subsurface (20-40 cm) profiles. Bulk density of each core increment was calculated as the oven-dry weight of rock and root-free soil divided by total increment volume estimated

from the sampling probe. For each core collected at a study site, pH of surface soils (0-20 cm) was determined using a digital pH meter submerged in a 1:1 soil:water mixture. Soil texture for each site was measured on a composite sample of lawn surface soils (0-20 cm) using a rapid method developed by Kettler et al. (2008). For sites where a nonturf core was substituted for a lawn core, texture measurements were made on a composite from both lawn and wild/native vegetation cover.

Farmland sites

Soil samples were collected from relict farmland in the Salt Lake Valley metropolitan area in September and October 2007. No effort was made to select sites randomly, due in part to the small number of agricultural fields present in the study area. Permission to sample was granted at three properties: two farms composed of multiple fields and a small retired pasture nestled within a dense residential area. At each of the larger sites, soils were collected from two fields having distinct crop covers. Two replicate cores were extracted from each field using the same coring methods described above for residential sites. The first core came from a random location in the field, and the second was taken 10 m northeast of the first core at an azimuth of 60 degrees. Preparation and analysis of individual depth increments mirrored that performed on residential soils. As a way of evaluating soil compaction effects associated with sampling with a small diameter probe, I also collected two cores from each farm property using a 5 cm diameter hand-impacting soil corer (AMS, Idaho Falls, ID). These larger cores were extracted either 1 m north or south of a corresponding small diameter sampling point. A paired *t*-test indicated no significant difference ($P < 0.05$) in SOC or bulk density between samples collected using the two methods. Nevertheless, only measurements

obtained using the small diameter probe were used in comparisons with residential data. In order to maintain sample independence, I averaged soil C by farm property and depth prior to additional statistical analysis.

Modeling regional lawn SOC

Historic and future changes in regional lawn SOC (1900-2030) were modeled using empirical relationships developed in this study. On a decadal time step, net gains in lawn SOC to 20 cm were calculated for each existing single-family residential parcel in Salt Lake County. These values were then summed to estimate total lawn SOC gains or losses for Salt Lake County at each time step.

$$\text{Total lawn SOC gain} = \sum((SOC_{current} - SOC_{initial}) * \text{lawn area}) \quad (2.2)$$

In Equation 2.2, $SOC_{current}$ is the amount of lawn SOC (g/m^2) to 20 cm predicted at the time step of interest (e.g. 1920). $SOC_{initial}$ is the amount predicted immediately following transition of a parcel to residential use. Both quantities were estimated from a regression model describing the relationship between lawn SOC of study sites and house age. SOC per square m of lawn was multiplied by the estimated lawn area of each parcel to derive total parcel SOC stocks to 20 cm. The method of estimating parcel lawn area is described below.

$$\text{Lawn area} = P_A * L_P \quad (2.3)$$

$$P_A = \text{parcel area - structure footprints - other impervious} \quad (2.4)$$

Where,

P_A = total area of pervious surfaces on the parcel

L_P = proportion of pervious area that is lawn

Two terms in Equations 2.3 and 2.4, parcel area and the footprint of primary structures, were known for each residential parcel in Salt Lake County. The other terms had to be approximated using averages derived from my study sites. The proportion of pervious surfaces occupied by lawn was visually estimated at each study parcel. Total area covered by impervious surfaces other than structures (e.g., sidewalks and driveways) was remotely characterized by digitizing features in a GIS using field notes and 0.3 m resolution aerial photography. Parcels larger than 0.5 hectare were not included in the model, since my methods would have likely greatly overestimated lawn cover on these properties. Parcels were also disregarded if impervious surfaces exceeded total parcel area.

Results

Lawn SOC

SOC content of residential lawn surface soils (0-20 cm) exhibited a positive relationship with yard age ($r^2 = 0.64$, $P < 0.0001$; Fig. 2.2). Values ranged from 837 g/m^2 for a 7-year-old home to 8747 g/m^2 for a home 94 years old. Along a linear trend line fitted to the data, organic C in the upper 20 cm of lawn soils increased 48 g m^{-2} yr^{-1} and more than tripled between parcels 7 and 100 years old.

Lawn SOC was highest in the upper soil profiles, as is characteristic of most terrestrial ecosystems, and decreased progressively with depth (Table 2.2). Average SOC storage of urban lawns older than 50 years was higher than that reported by Jobbágy and

Table 2.2 Mean soil organic carbon (SOC), soil inorganic carbon (SIC), and bulk density of the three primary ground cover types sampled.

		SOC (g/m^2)		SIC (g/m^2)		Bulk density (g/m^3)	
	n	Mean	S.D.	Mean	S.D.	Mean	S.D.
Residential lawn							
0-10 cm	38	2417	833	733	632	0.90	0.17
10-20 cm	34	1545	930	1049	924	1.09	0.23
20-30 cm	23	1385	967	1478	1105	1.25	0.26
30-40 cm	20	1113	638	1361	1057	1.33	0.36
Residential nonturf							
0-10 cm	33	2198	1003	600	516	0.83	0.23
10-20 cm	22	1912	1274	894	847	0.86	0.27
20-30 cm	16	1568	1179	1356	1934	1.10	0.31
30-40 cm	11	992	648	1705	2559	1.19	0.30
Agriculture							
0-10 cm	3	2050	1141	540	309	0.99	0.11
10-20 cm	3	1415	241	580	353	1.18	0.23
20-30 cm	3	1195	185	720	630	1.20	0.20
30-40 cm	2	672	98	1584	359	1.09	0.11

Two lawn cores and one nonturf core were generally collected from each residential site. 2-4 cores were collected from each agricultural site. The above values were obtained by averaging the mean SOC, SIC and bulk density of each site. Underground obstructions prevented collection of some soil samples. *N* is the number of sites where at least one core was successfully retrieved.

Jackson (2000) for temperate grasslands and deserts at both 0-20 and 20-40 cm depths (Fig. 2.3). For lawns younger than 50 years, average SOC at both depths was higher than reported for deserts, but lower than reported for grasslands. Trends related to yard age also varied within the soil profile. The relationship between SOC and yard age was strongest at 10-20 cm and weakened progressively with depth (Fig. 2.4). Rates of SOC increase predicted by depth-specific regression models followed the same pattern.

Within-site variability in lawn SOC was evaluated by comparing the two lawn core replicates collected from most residential parcels. This variability is illustrated graphically in Figure 2.5. In order to address the possibility that high SOC measurements at older sites resulted from random sampling of carbon-rich microsites, I calculated the difference in SOC between replicate samples by depth. Prediction intervals of the resulting normal distributions were used to calculate the range of variation expected for any two cores extracted randomly from a residential lawn within the study area. Ninety-five percent of the time, such differences were expected to be less than 1951 g/m^2 at 0-10 cm, less than 1307 g/m^2 at 10-20 cm, and less than 2819 g/m^2 for the two depths combined. These values are considerably smaller than the range of SOC values observed at the same depths among sites. A *t*-test performed on paired front and rear lawn samples showed no significant difference in SOC between front and rear lawns at the 5% level.

Differences in yard age explained 64% of the variation observed in site-averaged SOC of lawn surface soils, based on standard least squares linear regression (Table 2.3). For depths of 0-20 cm, regressions using elevation, percent sand or percent silt as the independent variable also exhibited slopes significantly different from zero, although these variables explained much less variation in lawn surface SOC among sites than did

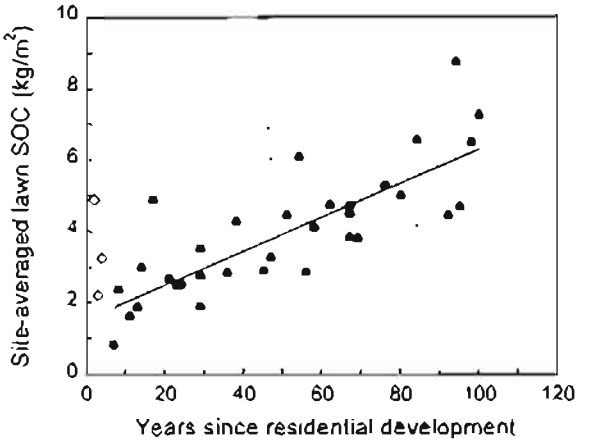

Fig. 2.2 Soil organic carbon (SOC) content of residential lawn surface soils (0-20 cm) as a function of yard age ($r^2 = 0.64$, $y = 0.048x + 1.6$). Each filled marker indicates the average value from replicate soil cores collected at a single site. SOC was calculated individually for 0-10 and 10-20 cm depth increments of each replicate core, averaged by depth increment, and then summed. For 18 of the 38 sites, one or more core increment could not be included in the calculation due to obstructions encountered while coring. Open markers indicate average SOC of farmland sites.

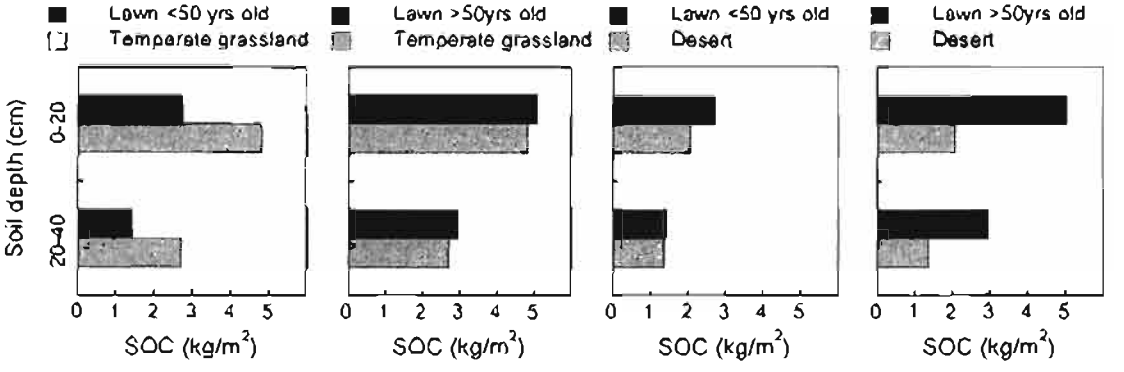

Fig. 2.3 Average soil organic carbon (SOC) of residential lawns, temperate grassland and desert. Values for grassland and desert come from Jobággy and Jackson (2000).

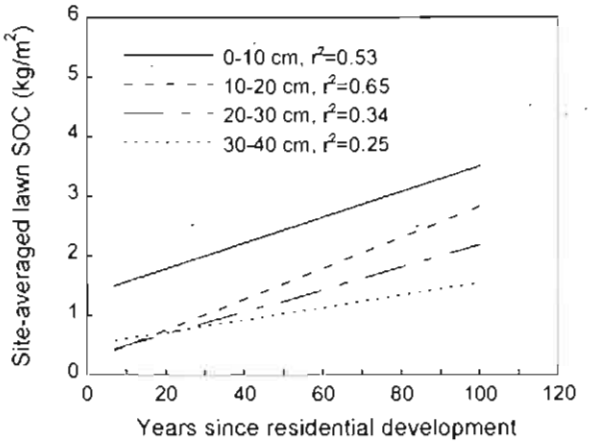

Fig. 2.4 Trends in lawn soil organic carbon (SOC) by depth. The relationship between site-averaged lawn SOC and yard age was evaluated using standard least squares regression.

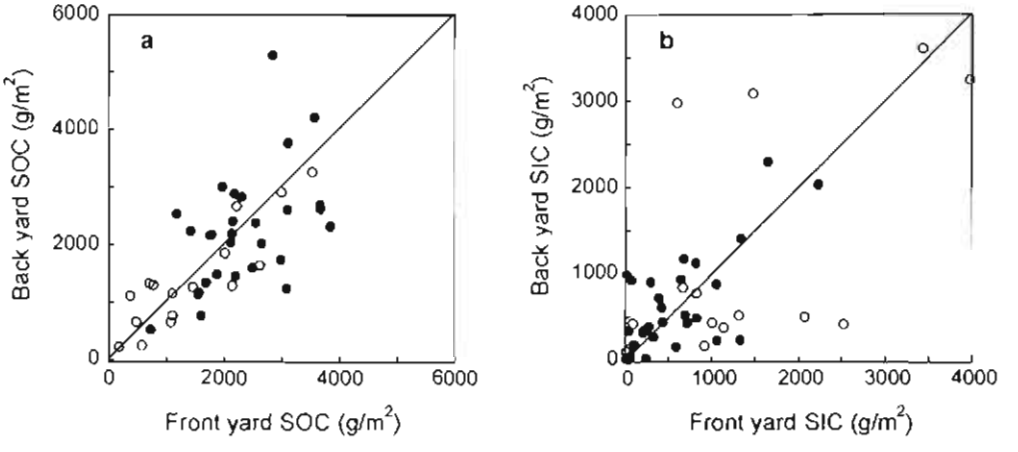

Fig. 2.5 Variability in **a** soil organic carbon (SOC) and **b** soil inorganic carbon (SIC) within residential lawns, as measured by replicate lawn soil cores. One core each was generally collected from the front and back yard. Filled markers represent a depth of 0-10 cm. Open markers represent 10-20 cm.

Table 2.3 Site characteristics regressed against average soil organic carbon (SOC) and soil inorganic carbon (SIC). R^2 is reported by soil depth (cm).

	Lawn SOC		Lawn SIC		Nonturf SOC		Nonturf SIC	
Variable	0-20	20-40	0-20	20-40	0-20	20-40	0-20	20-40
Age (yr)	0.64***	0.31*	0.01	0.01	0.19*	0.56**	0.01	0.19
Elevation (m)	0.31**	0.28*	0.21**	0.11	0.24*	0.10	0.20*	0.16
%Sand	0.19*	0.19	0.43***	0.30*	0.17	0.06	0.30**	0.26
%Silt	0.27**	0.26*	0.34**	0.23*	0.19*	0.12	0.23*	0.18
%Clay	0.02	0.03	0.48***	0.43**	0.08	0.00	0.34**	0.35
Age x elevation	0.73***	0.45**	0.21*	0.11	0.39**	0.64*	0.22	0.36
Age x %Sand	0.66***	0.38*	0.46***	0.31*	0.29*	0.57*	0.34*	0.54*

***$P < 0.0001$, **$P < 0.01$, *$P < 0.05$

yard age. Lawn SOC was positively related to percent silt and negatively related to elevation and percent sand. Multiple regressions combining yard age with each of the factors listed above only modestly increased performance. The best fit, made using yard age and elevation as independent variables, explained 73% of the variation in lawn SOC to 20 cm. The same variables also produced the best model for sub-surface soils (20-40 cm), although elevation had nearly equivalent explanatory power as yard age at this depth and overall performance was modest ($r^2 = 0.45$). Including soil texture did not improve either model due to a positive correlation between elevation and percent sand, and a negative correlation between yard age and percent sand, at my study sites. For those parcels developed later than 1959, SOC stocks (0-20 cm) of yards constructed on farmland were not significantly different from SOC stocks of yards built on undeveloped/rangeland vegetation, based on a least squares means Student's *t*-test performed on soil SOC when plotted against yard age. Soil bulk density exhibited no significant relationship to either yard age or lawn soil SOC.

Residential nonturf SOC

The relationship between surface SOC and yard age was considerably weaker for nonturf cover ($r^2 = 0.19$; $P = 0.0428$; Table 2.3). In general, surface SOC of nonturf ground cover exhibited high variability among sites, and regression analyses indicated no strong relationships between surface SOC and any one of the site characteristics evaluated. Linear regressions between SOC and soil texture variables were significant only at 10-20 cm ($P < 0.05$). With yard age as the only independent variable, performance of regression models generally improved with depth, and those using only elevation exhibited the opposite pattern. A multiple regression that included both age and elevation as independent variables explained 39% of the variation in surface nonturf SOC and 64% of the variation in subsurface nonturf SOC. These analyses were hampered by low sample size, particularly at depths of 30-40 cm.

Notably, despite the relatively high variability in nonturf surface SOC and weak relationship with age, the predicted increase in nonturf SOC over time followed a similar trend to that of lawn SOC (Fig. 2.6). Indeed, there was no significant difference between the regression slopes or least squares means of the two ground covers when site-averaged SOC was plotted as a function of yard age (Student's *t*-test, $P < 0.05$). The regression between site turf and nonturf SOC was nevertheless weak at 0-20 cm ($r^2 = 0.24$) and moderate at 20-40 cm ($r^2 = 0.40$).

Soil inorganic C

SIC content of urban soils was highly variable both between and within sites and exhibited no clear trend with yard age (Fig. 2.5; Table 2.3). Regression models using yard age as the independent variable were not significant for any depth. Across all residential

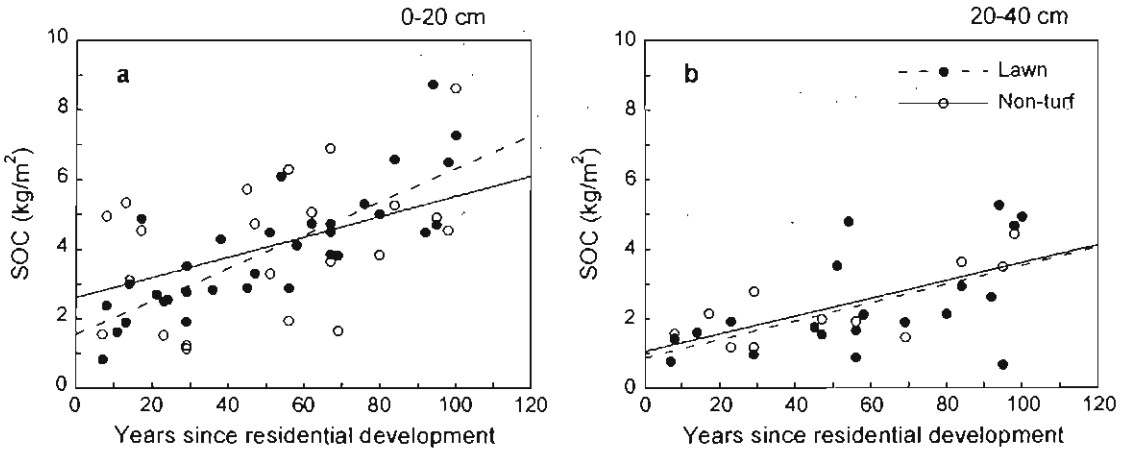

Fig. 2.6 Trends in residential lawn and nonturf soil organic carbon (SOC) as a function of yard age. Markers indicate average lawn and nonturf soil SOC for each study site at **a** 0-20 and **b** 20-40 cm. Trend-lines were derived using standard linear regression. In general two soil cores were collected from lawn at each site, and one core was collected from nonturf. No significant difference was observed between the least squares means or regression slopes of site-averaged lawn and nonturf SOC at either depth when plotted against yard age ($P < 0.05$).

sites, average SIC increased with depth beneath both lawn and nonturf ground cover (Table 2.2). Agricultural sites exhibited the same trend, although little change in average SIC occurred between 0 and 30 cm. Of the site characteristics examined, soil texture and elevation were most effective at explaining variation in SIC of urban soils. Percent clay exhibited the strongest relationship to SIC beneath both lawn and nonturf (Table 2.3; Fig. 2.7). Regression models using this variable were significant ($P < 0.05$) at all depths beneath lawn and in surface soils of nonturf. After removing four outliers from the dataset, I observed a slight negative trend in soil pH with yard age ($r^2 = 0.43$, $P < 0.0001$). Nevertheless, no relationship was apparent between pH and SIC content of urban soils.

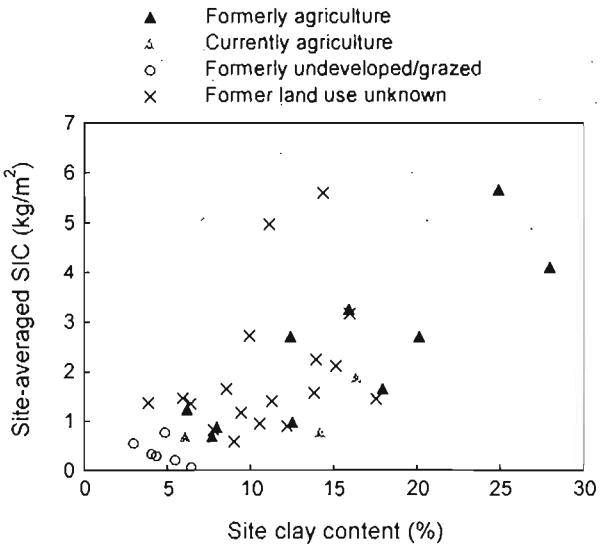

Fig. 2.7 Lawn soil inorganic carbon (SIC) to 20 cm as a function of clay content. Each marker indicates the average value from replicate soil cores collected at a single site. SIC was calculated individually for 0-10 and 10-20 cm depth increments of each replicate core, averaged by depth increment, and then summed. Prior land use (farmland or undeveloped/rangeland) was determined for all sites built later than 1959 using aerial photography.

Discussion

Site-level C stocks

Human-mediated land cover transformations have the potential to significantly alter soil C storage. This phenomenon is most well documented with regard to agricultural expansion, where tilling of native ecosystems has led to rapid C loss in many cases (Aguilar et al. 1988; Burke 1989; Davidson and Ackerman 1993). On the other hand, the large size of the global soil C pool has fueled optimism that soils might be managed for atmospheric C uptake (Schlesinger 2000; Lal 2004; Post 2004). A growing volume of studies suggest that urbanization of arid and semiarid regions in the US can lead to accumulation of organic C in soils beneath turfgrass and other urban vegetation. My results support these earlier findings. Across an urban age gradient spanning nearly a century, I observed a highly significant positive relationship between yard age and surface SOC of residential lawns (Fig. 2.2). Variability of SOC within individual lawns, although high at depths of 0-10 cm, is unlikely to explain the range of SOC observed across the age gradient, based on analysis of replicate samples collected from individual sites. I infer from these results that residential lawn soils are accumulating organic C in this semiarid region. At depths of 0-20 cm, rates of organic C gain may be as high as 48 g m^{-2} yr^{-1} and may persist near that rate for more than a century. These results are consistent with my original expectation that the high productivity of lawns and lack of annual disturbance should lead to increases in lawn SOC stocks in the study region.

The strong trend between lawn age and SOC content of surface soils occurred despite considerable variation in other site characteristics. Only elevation and soil texture emerged as additional factors having significant influence on lawn SOC (Table 2.3). With

regard to elevation, the negative relationship exhibited by my data is contrary to expectations based on trends of precipitation increase and temperature decrease typically associated with elevation gain. Sites in this study occupied a relatively narrow elevation range of 240 m. Nevertheless, mean annual temperature is about 0.4 degrees C cooler at a weather station near the upper bound of this range than at a weather station near the lower bound (WorldClimate 2008) Mean annual precipitation is reported to be 40-50% greater at the higher elevation station. The reduced SOC stocks found at my high elevation sites are presumably the result of differences in native soil characteristics, site history and/or hydrology. Supporting this conclusion is the lack of a significant relationship between yard age and elevation. The valley bottom of the study area is affected by alluvial deposits from the Great Salt Lake and Jordan River, while valley benches are largely formed through erosion of the adjacent mountains. These topographic differences contribute to a positive correlation between elevation and percent sand at these urban sites. In addition, the lowest elevations of the valley are only slightly above current levels of the Great Salt Lake, potentially experienced flooding in the recent past, and prior to urban development, may have had conditions favorable to higher plant productivity and SOC storage than well-drained upland sites. Elevation may thus offer a reasonable yardstick with which to estimate geographic differences in SOC stocks prior to urbanization of the Salt Lake City metropolitan area. This suggestion is supported by a series of soil pits sampled by the US Soil Survey along a transect spanning the full elevation range of this study. These samples were presumably collected beneath either farmland or natural vegetation, and data from them indicate a steady decrease in SOC with elevation (Soil Survey Staff 2008, data not shown).

Soil texture often exerts a strong control on SOC content in nonurban settings. Physical and chemical associations between organic matter and silt and clay-sized particles are known to protect organic matter against decay (Hassink 1997; Mikutta et al. 2006; Plante et al. 2006). As a result, SOC is positively related to clay content on global and regional scales (Nichols 1984; Burke et al. 1989; Jobbágy and Jackson 2000; Homann et al. 2007), although other factors may obscure this general pattern in certain locations (Sims and Nielsen 1986; Hontoria et al. 1999; Percival et al. 2000). I observed only weak relationships between soil texture and lawn SOC at my residential sites (Table 2.3). Because none of the oldest sites exhibited high sand content, my data cannot address the potential for low clay and silt content to constrain SOC storage at steady state. Nevertheless, the relatively poor performance of regression models using only soil texture or elevation as the independent variable suggests that variation in lawn surface SOC can be attributed primarily to anthropogenic influence (i.e., yard age). Irrigation and nutrient inputs likely reduce or obscure the influence of site factors commonly important in natural settings, at least in surface soils. These findings are consistent with predictions made by Pouyat et al. (2003) that SOC densities in US cities should converge on similar values despite differences in climate and vegetation prior to development. At depths below 20 cm, the amount of variability explained by yard age was similar to that explained by elevation or texture (Table 2.3). Indeed, a correlation between yard age and soil texture makes it difficult to differentiate the effects of these variables at these depths.

This study indicates that SOC may accrue beneath lawns in semiarid regions for a considerably longer time than previously reported. Golf course soil testing records analyzed by Qian et al. (2003) suggested that organic matter content of surface soils

tended to reach equilibrium about 30 to 45 years after golf course construction. CENTURY model simulations of golf course SOC dynamics in the same region produced similar results (Bandaranayake et al. 2003). Findings reported here point to the possibility of a much longer approach to steady state. Fitting a power function to the data shown in Figure 2.2 increased the coefficient of determination only slightly ($r^2 = 0.65$) and this curve reached its apex well beyond 100 years. It is possible that differences in initial SOC storage among my residential sites are responsible for the discrepancy between my results and those of previous studies. My methods may overestimate the duration and rate of lawn SOC increase if pre-urban stocks at the oldest sites were significantly higher than at other properties sampled. Unfortunately, I could not test this possibility, since the high density of urban development in the older portions of the Salt Lake Valley metropolitan area made it difficult or impossible to pair urban sites with representative sites supporting farmland and natural vegetation.

On average, lawns occupied about 41% of the ground surface at residential sites sampled in this study (Table 2.4). Pervious surfaces without turf occupied about 29%, and these areas consisted of a variety of vegetated and nonvegetated cover types. I anticipated changes in SOC over time to be greater beneath lawns than nonturf areas due to the characteristically high plant density and productivity of lawns. Instead, my limited sampling of nonturf areas indicated no significant difference in average SOC stocks beneath the two ground covers when plotted against yard age (Figure 2.6). At least two explanations may account for this finding. Residential yards can be highly fluctuating landscapes, and the transition of yard areas in and out of lawn cover may help explain the similar trends observed in lawn and nonturf SOC with respect to yard age. Alternatively,

Table 2.4 Mean percent cover of water pervious and impervious surfaces on residential study parcels ($n = 40$)

Ground cover type	Mean % cover
Pervious surfaces	69 (1.6)
Vegetated	
Maintained turf	41 (2.5)
Horticultural herb, ivy	6 (1.0)
Wild/natural vegetation	8 (2.2)
Nonvegetated	
Organic mulch, duff	5 (0.7)
Rock, gravel, brick, pavers	3 (0.5)
Bare soil	7 (0.2)
Impervious surfaces	31 (1.6)
Structures	17 (1.0)
Concrete, asphalt, mortared brick/pavers	14 (1.3)

Ground cover was visually estimated in the field. Cover estimates include areas beneath trees and shrubs. One standard deviation is given in parentheses.

organic mulches and soil amendments applied to ornamental gardens may result in accumulation of soil organic matter in these areas. Byrne et al. (2008) monitored soil properties of adjacent plots covered with mowed turfgrass, unmowed vegetation, bark mulch or gravel mulch. Three years after establishment, soils beneath lawn and bark mulch had slightly higher C concentrations than measured below unmowed vegetation and gravel. In contrast, Scharenbroch et al. (2005) observed no accumulation of SOM in soils covered by bark mulch for 15 years. Because the second explanation involves importation of C from outside the residential parcel, and presumably from outside the urban area, it would not represent direct C sequestration by residential yards. Turfgrass systems may also receive organic amendments, particularly in the form of imported

topsoil. While the extent of this practice is not known for my study area, it presumably would contribute minimal volume to most soil profiles, either as a thin layer of soil included with turfgrass sod or as a shallow addition to excessively rocky or poorly developed soils.

High variability in surface nonturf SOC among residential sites blurred relationships with yard age and elevation, and it resulted in SOC being nearly independent of soil texture. This large variation no doubt arose in part from my sampling design, which involved collecting only a single nonturf core from each site. Considerable diversity in the types of ground cover sampled also likely contributed to this variability. Because of high variability in soil inorganic C measurements within residential lawns, the number of replicates collected may not have been adequate to detect differences in SIC among sites (Fig. 2.5). Nevertheless, a consistent increase or decrease in SIC at the depths sampled seems unlikely over a century of residential use. Golubiewski (2006) also found no significant trend in lawn SIC among residential properties of various ages.

Regional C stocks of urban soils

Across undeveloped landscapes, SOC has been found to vary at a range of scales (Guo et al. 2006; Garten et al. 2007). Some of this variability can be explained by site conditions known to influence SOC dynamics, and some of it cannot (Jobbágy and Jackson 2000). The strong relationship observed here between lawn SOC and the age of residential yards implies that historic patterns of urban development may strongly influence the geographical distribution of SOC in urbanized regions of the semiarid US. Not only are older neighborhoods in the Salt Lake Valley metropolitan area likely to have higher SOC densities than newer neighborhoods, SOC content may also vary predictably

from one parcel to another. The abrupt and often linear boundaries between properties suggest that SOC storage may be highly discontinuous across this and other urban ecosystems. Legal parcels may thus form a useful unit of study for regional and continental evaluations of urban SOC storage.

My results indicate that temporal patterns of urban development may also influence current SOC inventories of some cities. All else being equal, an aridland city consisting predominantly of new development would likely have lower SOC inventories than a nearby city in which the majority of neighborhoods are many decades old. As a way of illustrating the importance of history on urban SOC stocks in my study area, I constructed a simple model to predict the historic and future changes in total C storage of surface soils beneath Salt Lake Valley residential lawns. This model indicated only small gains in lawn SOC stocks over the early half of the past century, due to the fact that most development occurred later than the 1940s (Fig. 2.8). The model predicted an exponential increase in lawn SOC accumulation during the latter half of the twentieth century, following an exponential rise in single-family homes in Salt Lake County.

A second, perhaps more realistic model scenario took into account evidence suggesting initial decline in surface SOC following house and yard construction. SOC stocks measured at agricultural sites in this study were notably higher than average SOC of young lawns, based on regression models pairing SOC with lawn age (Fig. 1.2). Young lawns in the Denver, CO metropolitan area also exhibited lower SOC stocks than nearby native grassland (Golubiewski 2006). Both findings suggest that ground disturbances during home and yard construction often initially reduce surface SOC stocks following development. Such an outcome is expected if surface soils are mixed with

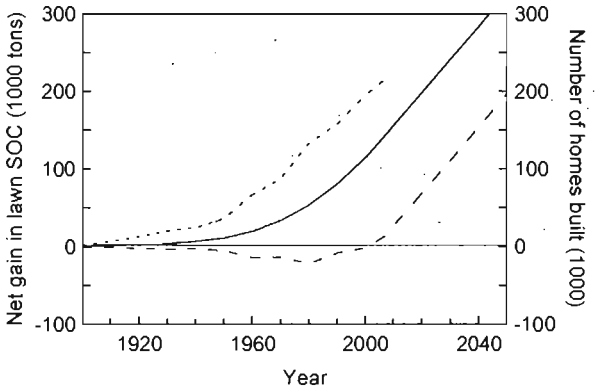

Fig. 2.8 Modeled historic and future gains in regional soil organic carbon (SOC) beneath Salt Lake Valley residential lawns (solid line). Only the upper 20 cm of the soil profile was included. The small dashed line indicates the number of single-family homes existing in Salt Lake County at the start of each decade. The large dashed line shows modeled changes in regional lawn SOC assuming initial losses of SOC during home and yard construction.

deeper profiles having lower organic C content. SOC losses might also occur from disruptions to soil aggregates during construction activities, resulting in exposure of physically protected organic matter to microbes, or from the "priming" of decomposition processes by increased inputs of high quality turfgrass litter (Fontaine et al. 2007). When the above model was adjusted to include estimated average initial SOC stocks of 3000 g/m^2, predicted SOC gains in the upper 20 cm were slightly negative for all decades between 1900 and 2000. After this time SOC stocks finally rose beyond estimated pre-urban conditions. Both model scenarios predicted that the largest gains in regional lawn SOC would occur in the current century.

Conclusions

Urbanization in the US is often characterized by extensive modification of existing ecosystems. Such modifications may include the introduction of novel landscape features

such as buildings and roads, as well as vegetation changes tied to landscaping trends and the preferences of individual urban residents. Changes in vegetation may result in changes in the biogeochemical properties of soils (Schlesinger 2000). Such a phenomenon is evident from SOC measurements collected here along an urban development age gradient spanning nearly a century in the semiarid Salt Lake Valley, UT. Predicted SOC of residential lawns to 20 cm was more than three-fold greater at the oldest sites than the youngest sites, based on standard linear regression. For a residential lawn of average size, this century-long accumulation amounts to roughly 22% of equivalent CO_2 (CO_2e) emissions currently produced by a single Utah resident in one year (Roe et al. 2007). CO_2e includes all greenhouse gas emissions expressed in terms of CO_2 based on the radiative forcing of each gas relative to CO_2. Carbon uptake by lawn soils no doubt offsets an even smaller fraction of anthropogenic greenhouse gas emissions when emissions arising from lawn mowing, clipping disposal, fertilization and irrigation are taken into consideration (Jo and McPherson 1995, Kaye et al. 2004). In addition, changes in C stocks beneath residential lawns may not accurately represent net C sequestration associated with urbanization if site preparation results in initial losses of C to the atmosphere (Golubiewski 2006). The benefits of C sequestration must also be weighed against the use of fertilizer, herbicides and limited freshwater resources to maintain turfgrass in an unfavorable climate.

At a regional scale, this study demonstrates that urban landscaping choices can greatly influence the organic C content of surface soils. Maintenance of highly productive turfgrass and other horticultural flora in this semiarid region appears to significantly increase surface SOC following conversion of farmland and natural vegetation to mesic

residential yards. The amount of intersite variability in lawn surface SOC explained by native site characteristic such as soil texture or elevation was less than half that explained by yard age. The results presented here indicate that accumulation of SOC beneath lawns in the Salt Lake Valley occurs predominantly in the upper 20 cm. They also suggest that long-term changes in SOC beneath pervious ground cover types other than turfgrass are similar to, but more variable than, those occurring beneath lawn. The significant relationship observed between SOC storage and time since residential development suggests that the geographic variability of SOC within the Salt Lake Valley is greatly influenced by historic patterns of urban development. Approaches that take into account the current age distribution of urban land cover are likely to increase the accuracy of urban SOC estimates in arid and semiarid regions.

References

Aguilar R, Kelly EF, Heil RD (1988) Effects of cultivation on soils in northern Great Plains rangeland. Soil Science Society of America Journal 52:1081-1085

Austin AT, Vitousek PM (1998) Nutrient dynamics on a precipitation gradient in Hawai'i. Oecologia 113:519-529

Bandaranayake W, Qian YL, Parton WJ, Ojima DS, Follett RF (2003) Estimation of soil organic carbon changes in turfgrass systems using the Century model. Agronomy Journal 95:558-563

Burke IC, Yonker CM, Parton WJ, Cole CV, Flach K, Schimel DS (1989) Texture, climate, and cultivation effects on soil organic matter content in U.S. grassland soils. Soil Science Society of America Journal 53:800-805

Byrne LB, Bruns MA, Kim KC (2008) Ecosystem properties of urban land covers at the aboveground-belowground interface. Ecosystems DOI: 10.1007/s10021-008-9179-3

Davidson EA, Ackerman IL (1993) Changes in soil carbon inventories following cultivation of previously untilled soils. Biogeochemistry 20:161-193.

Eshel G, Fine P, Singer MJ (2007) Total soil carbon and water quality: an implication for carbon sequestration. Soil Science Society of America Journal 71:397-405

Eswaran H, Reich PF, Kimble JM, Beinroth FH, Padmanabhan E, Moncharoen P (2000) Global carbon stocks. In Lal R, Kimble JM, Eswaran H, Stewart BA (eds) Global climate change and pedogenic carbonates. Lewis Publishers, Boca Raton London New York Washington, DC, pp 15-25

Fontaine S, Barot S, Barré, Bdioui N, Bruno M, Rumpel C (2007) Stability of organic carbon in deep soil layers controlled by fresh carbon supply. Nature 450:277-281

Garten CT Jr, Kang S, Brice DJ, Schadt CW, Zhou J (2007) Variability in soil properties at different spatial scales (1 m-1 km) in a deciduous forest ecosystem. Soil Biology and Biochemistry 39:2621-2627

Golubiewski NE (2006) Urbanization transforms prairie carbon pools: effects of landscaping in Colorado's Front Range. Ecological Applications 16:555-571

Guo Y, Amundson R, Gong P, Yu Q (2006) Quantity and spatial variability of soil carbon in the conterminous United States. Soil Science Society of America Journal 70:590-600

Hassink J (1997) The capacity of soils to preserve organic C and N by their association wilt clay and silt particles. Plant and Soil 191:77-87

Homann PS, Kapchinske JS, Boyce A (2007) Relations of mineral-soil C and N to climate and texture: regional differences within the conterminous USA. Biogeochemistry 85:303-316

Homer C, Dewitz J, Fry J, Coan M, Hossain N, Larson C, Herold N, McKerrow A, VanDriel JN, Wickham J (2007) Completion of the 2001 National Land Cover Database for the conterminous United States. Photogrammetric Engineering and Remote Sensing 73:337-341

Hontoria C, Rodríguez-Murillo JC, Saa A (1999) Relationships between soil organic carbon and site characteristics in peninsular Spain. Soil Science Society of America Journal 63:614-621

Jenny, H (1941) Factors of soil formation. McGraw-Hill Book Company, Inc., New York London

Jo H, McPherson EG (1995) Carbon storage and flux in urban residential greenspace. Journal of Environmental Manage 45:109-133

Jobággy EG, Jackson RB (2000) The vertical distribution of soil organic carbon and its relation to climate and vegetation. Ecological Applications 10:423-436

Kaye JP, Burke IC, Mosier AR, Guerschman JP (2004) Methane and nitrous oxide fluxes from urban soils to the atmosphere. Ecological Applications 14:975-981

Kaye JP, McCulley RL, Burke IC (2005) Carbon fluxes, nitrogen cycling, and soil microbial communities in adjacent urban, native and agricultural ecosystems. Global Change Biology 11:575-587

Kettler TA, Doran JW, Gilbert TL (2008) Simplified method for soil particle size determination to accompany soil-quality analyses. Soil Science Society of America Journal 65:849-852

Khokhlova OS, Arlashina EA, Kovalevskaya IS (1997) The effect of irrigation on the carbonate status of Chernozems of Central Precaucasus (Russia). Soil Technology 11:171-184

Lal R (2004) Soil carbon sequestration impacts on global climate change and food security. Science 304:1623-1627

Mann LK (1986) Changes in carbon storage after cultivation. Soil Science 142:279-288

Mikutta R, Kleber M, Torn MS, Jahn R (2006) Stabilization of soil organic matter: association with minerals or chemical recalcitrance? Biogeochemistry 77:25-56

Milesi C, Running SW, Elvidge CD, Dietz JB, Tuttle BT, Nemani RR (2005) Mapping and modeling the biogeochemical cycling of turf grasses in the United States. Environmental Management 36:426-438

Nowak DJ, Crane DE (2002) Carbon storage and sequestration by urban trees in the USA. Environmental Pollution 116:381-389

Nichols JD (1984) Relation of organic carbon to soil properties and climate in the southern Great Plains. Soil Science Society of America Journal 48:1382-1384

Percival HJ, Parfitt RL, Scott NA (2000) Factors controlling soil carbon levels in New Zealand grasslands: Is clay important? Soil Science Society of America Journal 64:1623-1630

Plante AF, Conant RT, Stewart CE, Paustian K, Six J (2006) Impact of soil texture on the distribution of soil organic matter in physical and chemical fractions. Soil Science Society of America Journal 70:287-296

Post WM, Izaurralde RC, Jastrow JD, McCarl BA, Amonette JE, Bailey VL, Jardine PM, West TO, Zhou J (2004) Enhancement of carbon sequestration in US soils. BioScience 54:895-908

Post WM, Pastor J, Zinke PJ, Stangenberger AG (1985) Global patterns of soil nitrogen

storage. Nature 317:613-616

Post WM, Mann LK (1990) Changes in soil organic carbon and nitrogen as a result of cultivation. In Bouwman AF (ed) Soils and the greenhouse effect. John Wiley and Sons Ltd., West Sussex, England pp 401-406

Pouyat RV, Russell-Anelli J, Yesilonis ID, Groffman PM (2003) Soil carbon in urban forest ecosystems. In Kimble et al. JM (eds) The potential of U.S. forest soils to sequester carbon and mitigate the greenhouse effect. CRC Press, Boca Raton, pp 347-362

Qian Y, Follet RF (2002) Assessing soil carbon sequestration in turfgrass systems using long-term soil testing data. Agronomy Journal 94:930-935

Roe S, Strait R, Bailie A, Lindquist H, Jamison A (2007) Final Utah greenhouse gas inventory and reference case projections, 1900-2020. Center for Climate Strategies, Utah Department of Environmental Quality

Salt Lake County Assessor (2008) Salt Lake County Parcel database.

Scharenbroch BC, Lloyd JE, Johnson-Maynard JL (2005) Distinguishing urban soils with physical, chemical, and biological properties. Pedobiologia 49:283-96

Schlesinger WH (1991) Biogeochemistry: an analysis of global change. Academic Press, Inc., San Diego

Schlesinger WH (2000) Carbon sequestration in soils: some cautions amidst optimism. Agriculture, Ecosystems and Environment 82:121-127

Sherrod LA, Dunn G, Peterson GA, Kolberg RL (2002) Inorganic carbon analysis by modified pressure-calcimeter method. Soil Science Society of America Journal 66:299-305

Sims ZR, Nielsen GA (1986) Organic carbon in Montana soils as related to clay content and climate. Soil Science Society of America Journal 50:1269-1271

Soil Survey Staff (2007) Soil Survey Geographic (SSURGO) Database for Salt Lake County, UT. Natural Resources Conservation Service, USDA. http://soildatamart.nrcs.usda.gov

Soil Survey Staff (2008) National Soil Survey Characterization Data. Soil Survey Laboratory, National Soil Survey Center, USDA-NRCS, Lincoln, NE

Tan KH (1996) Soil sampling, preparation and analysis. Marcel Dekker, Inc., New York

US Census Bureau (1952) U.S. census of agriculture: 1950. Counties and state economic areas, part 31. US Government Printing Office, Washington, DC

Wagner SW, Hanson JD, Olness A, Voorhees WB (1998) A volumetric inorganic carbon analysis system. Soil Science Society of America Journal 62:690-693

Wakefield JH (1933) A study of the plant ecology of the Salt Lake and Utah Valleys before the Mormon immigration. MS thesis, Brigham Young University, 56 pp

Wall D, Anderson DL, Austin LH, Gillette P, Strong D, Stauffer N, Summers L, Klotz E, Adams T, Williams K, Austin A (2001) Identifying residential water use: survey results and analysis of residential water use for thirteen communities in Utah. Utah Department of Natural Resources, Division of Water Resources, 56 pp

WorldClimate (2008) http://www.worldclimate.com

CHAPTER 3

SOIL NITROGEN STOCKS ALONG AN URBAN AGE GRADIENT IN SEMIARID SALT LAKE VALLEY, UT

Abstract

Lawns are a dominant urban land cover in the USA, often receiving large quantities of synthetic nitrogenous fertilizer and heightened atmospheric-N deposition. A portion of these N inputs may be sequestered in lawn soil. Long-term N accumulation beneath lawns and other pervious surfaces was examined along an urban development age gradient in the Salt Lake Valley, USA metropolitan area. Soils were sampled from 38 single-family residential parcels ranging in age from 7 to 100 years. Linear regression showed a highly significant positive relationship between time since development and total nitrogen (TN) stocks of surface soils (0-20 cm) beneath residential lawns ($r^2 = 0.64$). The slope of the regression suggests that average lawn surface soil TN in this urban ecosystem increases at a rate of about 3.9 g m^{-2} yr^{-1}. Lawn soil below 20 cm exhibited a slower rate of increase, as did residential nonturf soils at all depths. Soil TN and organic carbon stocks were strongly correlated. The results presented here point to regional differences in the capacity of lawn soils to retain applied N. In the semiarid Salt Lake Valley, low initial soil organic matter stocks may allow N to accrue for longer than a century. Among residential sites, surface bulk soil $\delta^{15}N$ increased significantly as a

function of lawn age. This trend was subordinate to a relationship between bulk soil $\delta^{15}N$ and soil texture of residential lawns.

Introduction

Human activities have greatly increased nitrogen inputs to the terrestrial biosphere (Vitousek et al. 1997), much of which is inadvertently exported to nontarget systems (Howarth et al. 2002). Unintentional N fertilization downwind or downstream of N sources has been shown to stimulate primary production in both terrestrial and aquatic ecosystems (Howarth 1988; Elser et al. 1990; Vitousek and Howarth 1991). Chronic inputs to terrestrial systems may promote increased nitrification, loss of nitrate (NO_2) through leaching and runoff, altered nutrient content of plant tissue and soil, and changes in plant community composition (Vitousek et al. 1997; Fenn et al. 1998). Notable exports of nitrous oxide (N_2O), a gas having 300 times the global warming potential of CO_2, have also been linked to fertilizer and irrigation application in agricultural and horticultural settings (Hutchinson and Mosier 1979; Mosier et al. 1991, Kaye et al. 2004; Hall et al. 2008).

Retention of N in soils is an important process mediating an ecosystem's response to increased N inputs. Studies have demonstrated that many ecosystems have considerable capacity to store applied N, a trait that may reduce N losses in gaseous and soluble forms (e.g. Starr and DeRoo 1981; Delgado et al. 1996; Nadelhoffer et al. 1999; Johnson et al. 2000). Higher N retention is expected for ecosystems in which N is a limiting resource, especially where N has been removed from the system by harvesting or natural disturbance (Aber et al. 1998). Goodale and Aber (2001) have demonstrated that land use modifications occurring more than a century previous may influence N cycling

and storage in forest ecosystems. Because organic matter fuels microbial immobilization and stabilizes N in pools with relatively slow turnover, soils that are accumulating organic matter are more likely to retain N following deliberate or inadvertent fertilization (Fenn et al. 1998).

This chapter reports on trends in soil N storage along an urban age gradient in the Salt Lake Valley, UT metropolitan area. Human activities influence a number of ecological processes in urban landscapes, including N cycling (Baker et al. 2001; Zhu et al. 2006). Some of these impacts, such as the application of N fertilizers to lawns, may be intentional. Others, like production of nitric oxide (NO) during the burning of fossil fuels, may not. Turfgrass is a prominent urban land cover in the US, and soil retention of applied N may figure prominently into the N cycle of turfgrass systems (Fig. 3.1). Not only are N inputs to lawns typically high. Studies from a variety of regions have observed accumulation of soil organic matter (SOM) following turfgrass establishment (Qian and Follett 2002; Golubiewski 2006; Shi et al. 2006). This phenomenon suggests the potential for soil organic N stocks beneath lawns to increase over time, with a portion of anthropogenic N inputs being first incorporated into plant tissue and microbial biomass and products, and subsequently deposited into a SOM pool with relatively slow N turnover (Raciti et al. 2008).

Organic N accumulation is expected to saturate in undisturbed lawn soils as organic matter input and decomposition approach a new equilibrium (Porter et al. 1980). This decline in N storage over time may have important implications for turfgrass management. Specifically, Porter et al. (1980) noted that reduced capacity of older lawns to retain N may lead to heightened N losses to the atmosphere and groundwater if

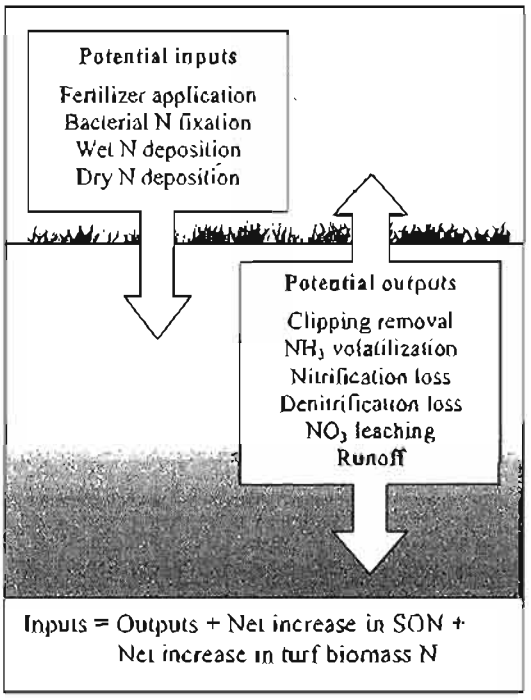

Fig. 3.1 Nitrogen balance of a typical turfgrass/soil system. Inputs are equal to the sum of outputs, net increase in soil organic N (SON) and net increase in turf biomass N. Adapted from Porter et al. (1980).

fertilizer application is not adjusted accordingly. Their study of long-term soil organic N dynamics beneath turfgrass in New York, USA suggested that average N accumulation nearly ceased after about 30 years.

Here, I examined soil N accumulation beneath residential yards in a highly urbanized region of the semiarid US. Since long-term monitoring of soil N would require decades, I attempted to quantify temporal trends in urban soil N stocks by substituting space for time. Soil samples were collected from lawn and other pervious ground cover at 38 single-family properties together representing each decade of development from 1907 to 2000. Relationships between soil N stocks and yard age were used to infer both the magnitude and duration of soil N accumulation beneath residential yards. This research was initiated under the premise that urbanization in the arid and semiarid US should differ in its impacts on soil biogeochemistry than urban development in more humid regions. First, the population of the western US is highly metropolitan. Only the northeastern US has a greater proportion of people living in cities (Hobbs and Stoops 2002). Clustering of the population in urban areas results in high deposition of anthropogenic N within and around urban centers and relatively low deposition elsewhere (Fenn et al. 2003). In these regions, mountains can play a notable role in directing and restricting the distribution of airborne N compounds (Fenn et al. 2003; Kammerdiener 2004). Secondly, soil organic matter stocks are typically low in temperate arid and semiarid ecosystems (Jobbágy and Jackson 2000). Urbanization often replaces these ecosystems with highly productive turfgrass and other vegetation common to wetter climates. While SOM stocks of urban soils are predicted to converge on similar values regardless of geography and climate (Pouyat et al. 2003), the lower initial SOM stocks of

arid and semiarid regions hints at a greater capacity to retain anthropogenic N inputs in urban soils (Hall et al. 2008).

Additional insights into long-term N saturation of urban soils may come from the study of N stable isotopes. This approach offers an integrated measure of past N cycling processes (Robinson 2001), making it less sensitive to the high temporal variability commonly associated with leaching and gaseous N losses from lawn soil (Horgan et al. 2002; Easton and Petrovic 2004; Guillard and Kopp 2004; Bijoor et al. 2008). Discrimination against the heavier stable isotope, ^{15}N, occurs during N transformations by soil microorganisms, specifically during nitrification and denitrification (Högberg 1997). This isotope fractionation results in source pools that are ^{15}N enriched and N products that are ^{15}N depleted. Predicted increases in nitrification, denitrification and NO_3 leaching associated with N saturation of terrestrial ecosystems should result in heightened loss of depleted forms of nitrogen from the soil-plant system (Aber et al. 1998), potentially leading to detectable trends in plant and bulk soil N isotope ratio (Högberg 1990; Högberg et al. 1992; Högberg and Johannison 1993). Analysis of N stable isotope natural abundance has been proposed as a relatively simple means of assessing N saturation status and potential across broad geographic areas (Pardo et al. 2002; Amundson et al. 2003, Pardo et al. 2006). Here, I report trends in soil ^{15}N natural abundance among residential sites of varying ages, and I examine the utility of this measurement for clarifying the magnitude and duration of N accumulation in urban soils.

Materials and methods

Study sites

Salt Lake Valley, UT lies at the center of a major urban corridor nestled against the western flanks of the Wasatch Range. Mountains restrict urban development and the dispersion of atmospheric pollutants on both the east and west sides of the valley. The valley bottom is characterized by a semiarid climate with mean annual precipitation of 397 mm and mean average temperature of 11 degrees C (WorldClimate 2008). Despite unfavorable natural conditions, turfgrass and trees are abundant within the urbanized portions of the study area, where they are maintained and/or established through irrigation. Early writings describe the area as primarily grassland prior to European settlement in the 1840s, with trees generally confined to riparian corridors and upland shrubs being much less dominant than in modern times (Wakefield 1933).

Soil samples were collected from residential sites between May and November of 2007. I sampled only single-family residential parcels, which in 2007 constituted approximately 39% of urban land cover within the study area. Potential study sites were randomly selected from a Salt Lake County parcel dataset using a geographic information system (GIS). Initial selection was stratified by decade of development, with the age of the current home used as a proxy for the date of transition to residential land use. Due to the time required to collect tree information on large parcels (data not reported here, but concurrently obtained for related research) parcels larger than 0.25 hectares were rejected; all other single-family residential parcels in the county had a chance of being chosen. Random selection of parcels developed later than 1979 was also stratified by prior land use (irrigated agriculture, dryland agriculture, or range/wildland), as

determined from historic aerial photography. Letters were sent to the owners of selected parcels soliciting permission to access their property. Permission was granted for a total of 40 parcels representing all decades between 1900 and 2007. These sites were distributed throughout the Salt Lake Valley metropolitan area (Fig. 3.2). The maximum distance between sites was 33 km and the elevation range was 240 m. Aerial photography taken in 1958 was used to determine land use at sites prior to urban development. Of sites developed since 1959, 11 were constructed on farmland and 8 replaced undeveloped land or range. Soils could not be collected from one site due to time constraints, and another site had soils too shallow to adequately sample.

Soil sampling

Three soil cores were collected from each parcel to a depth of 40 cm. These generally included two randomly placed lawn cores (one core each from front and rear lawn, when possible) and a third core taken arbitrarily from a nonturf area such as flowerbed, shrub row or vegetable garden. A 2 cm diameter soil recovery probe (AMS, Idaho Falls, ID) allowed replicate samples to be obtained with minimal disturbance to homeowners' manicured yards. The total number of cores collected in the study is listed in Table 3.1 by ground cover type. Most residential lawns were dominated by Kentucky Bluegrass (*Poa pratense*), while an assortment of plant species, litter and mulches characterized the various yard features sampled as nonturf. Deviations from the study design described above include one site where no nonturf core was collected, three sites where a nonturf core was substituted for a lawn core, and two sites where an additional lawn core was substituted for a nonturf core.

Cores were extracted in 10 cm increments, placed in individual plastic bags, and

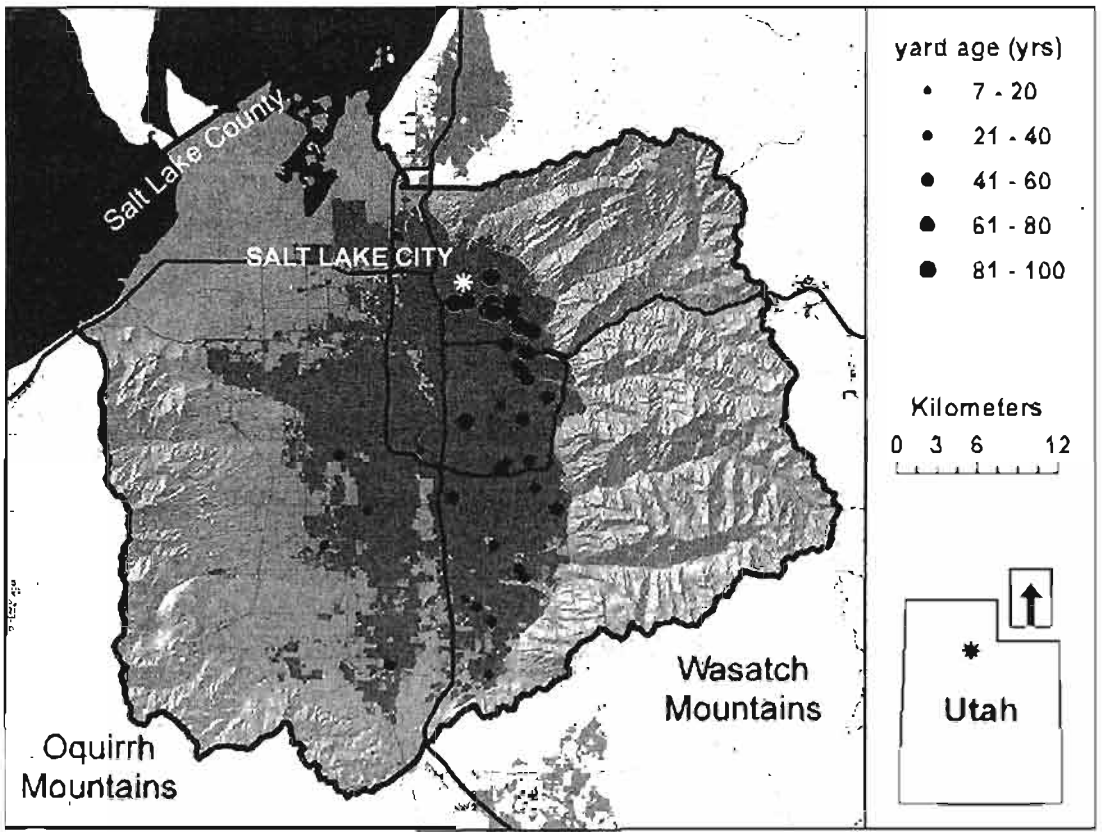

Fig. 3.2 Salt Lake Valley, USA and study sites locations. Larger circles indicate older residential sites, based on the age of the current home. Urban land cover from the National Land Cover Dataset 2001 is shown in medium grey (Homer et al. 2007).

Table 3.1 Ground cover at soil sampling points

Ground cover	No. of cores
Residential turf	74
Front lawn	44
Rear lawn	30
Residential nonturf	35
Horticultural forbs/shrubs	19
Wild/native vegetation	7
Bare soil	3
Organic mulch/duff (<5% plant cover)	3
Vegetable garden	3

transported back to the lab to be air dried and sieved (2 mm). Rocky soils prevented extraction of one or more increment at the majority of sites. Soil samples containing at least 70% of a full increment were retained and the depth recorded. Otherwise, I returned partial increments to the excavation and made no further effort to collect a complete core. The percentage of samples successfully retrieved dropped considerably with depth (96% of 0-10 cm increments, 68% of 10-20 cm increments, 40% of 30-40 cm increments, and 33% of 30-40 cm increments). Others have noted the challenges of sampling soils in an urban setting (Jo and McPherson 1995; Golubiewski 2006).

Soil analysis

Each 10 cm core increment was individually processed and analyzed. During sieving, increments were cleaned of visible roots using tweezers, and fine soil (<2 mm), coarse material (>2 mm), and fine root fractions (<2 mm diameter) were weighed. Two 5 g subsamples were isolated from air-dried and sieved soil through quartering (Tan 1996). One subsample was ground on a ball mill (MM200, Retch, Haan, Germany) for chemical

analysis, and the other was dried in an 80 degree Celsius forced-air oven for 48 hours to derive a moisture correction factor for soil texture and bulk density calculations. Percent total C, percent total N, and $\delta^{15}N$ were measured on 30 (±1) mg of air-dry and ground soil using an elemental analyzer coupled to an isotope ratio mass spectrometer (Finnigan MAT delta S, Thermo Electron Co, San Jose, CA). Soil reference material included in the analyses exhibited average standard deviations of 0.02, 0.00, and 0.4 for %C, %N, and $\delta^{15}N$(‰), respectively ($n = 40$). Inorganic C was measured on 300 (±10) mg of air-dried and ground soil via pressure calcimeter (Wagner et al. 1998, Sherrod et al. 2002), and organic C was determined by difference. Total N and organic C concentrations were converted to mass per unit ground surface area, integrated across each 10 cm depth, using the equation

$$TN\ (g\ m^{-2}) = \%TN/100 * F * (10\ cm/D)/A * 10\ 000 \quad (3.1)$$

where F is the fine fraction weight (g), D is the increment depth recovered (cm), and A is the cross-sectional area of the sampling probe tip (cm^2). Because this calculation does not include stones larger than 2 cm, it likely overestimates soil N content in most cases.

Bulk density of each core increment was calculated as the oven-dry weight of rock and root-free soil divided by soil volume estimated from the sampling probe. For each core collected at a study site, pH of surface soils (0-20 cm) was determined using a digital pH meter submerged in a 1:1 soil:water mixture. Soil texture for each site was measured on a composite sample of lawn soils (0-20 cm) using a rapid method developed by Kettler et al. (2008). Briefly, 10 g soil samples were each combined with 100 mL 3% sodium hexametaphosphate (HMP) solution and mechanically shaken for 2-3 hours. Following particle dispersion, the sand fraction was isolated by rinsing the soil solution

through a 0.053 mm sieve, and the silt fraction was isolated through sedimentation. Percent sand and silt were calculated using the oven-dry weight of the respective fraction and the oven-dry weight of the original sample. Subtracting percent sand and silt from 100 produced a value for percent clay. Organic matter was not removed prior to dispersion. This detail and the small quantity of soil used suggest that the protocol may slightly overestimate sand and underestimate clay. For sites where a nonturf core was substituted for a lawn core, texture measurements were made on a composite from both lawn and wild/native vegetation cover.

Statistical treatment

All statistical analyses were performed using JMP™ software. Relationships between soil TN and study site characteristics were examined using standard least squares linear regression. Nitrogen content, $\delta^{15}N$ and C:N ratio of replicate cores collected from the same site and ground cover were averaged by depth prior to regression analyses. Relationships having slopes significantly different from zero were included in a step-wise linear regression to identify models explaining the greatest amount of soil TN variation among residential sites. The same analyses were performed using $\delta^{15}N$ and C:N ratio as the dependent variable. Differences in soil $\delta^{15}N$ between lawn and nonturf cover were examined using the Student's *t*-test at 0-10 and 10-20 increments. At deeper increments, a nonparametric Wilcoxon test was employed due to lack of normality and unsuccessful use of data transformations. Student's *t*-test was also used to compare C:N ratio between lawn and nonturf cover.

Results

Lawn soil TN

Across the urbanization gradient sampled in this study, soil TN content increased as a function of yard age. The relationship was stronger for lawn surface soils (0-20 cm) than for surface soils beneath nonturf cover. Yard age alone explained 64% of the variation in lawn surface soil TN observed among sites, based on simple linear regression ($P < 0.0001$; Table 3.2). Average TN to 20 cm increased at a rate of 3.9 g m^2 yr^{-1} along this linear trend-line, with 10-year-old lawns having a predicted value of 193 g/m^2 and 100-year-old lawns having a predicted value of 547 g/m^2 (Fig. 3.3). The predicted rate of N increase was 1.9 g m^2 yr^{-1} for 0-10 cm and 2.0 g m^2 yr^{-1} for 10-20 cm. Below 20 cm, both the slope and performance of regressions diminished with soil depth, although relationships between soil TN and yard age were still significant at 20-30 and 30-40 cm ($P < 0.05$).

Site-averaged lawn soil TN exhibited significant but weaker relationships to elevation and soil texture in the top 20 cm ($P < 0.05$; Table 3.2). When percent silt or percent sand was paired with yard age in a multiple linear regression, these independent variables together explained 67% of the intersite variation in soil TN to 20 cm. Soil TN was negatively correlated to percent sand and positively correlated to percent silt. A regression using yard age and elevation as independent variables explained even more of the intersite variation in surface lawn soil TN ($r^2 = 0.75$). Including both elevation and soil texture only slightly improved model performance and led to problems of collinearity, since elevation and soil texture were correlated among study sites. Soil TN stocks of yards constructed on farmland were not significantly different from soil TN

stocks of yards replacing undeveloped/rangeland vegetation, based on a least squares means Student's *t*-test performed on soil TN when plotted against yard age. Soil bulk density exhibited no significant relationship to either yard age or lawn soil TN.

Residential nonturf soil TN

Nonturf soil TN generally exhibited a positive trend with yard age across the residential gradient sampled in this study, but this trend was characterized by considerable variability among sites, particularly in surface soils. Below 20 cm, statistical analyses suffered from low sample size. Nonetheless, the relationship between nonturf soil TN and yard age was significant at all depths below 10 cm and became progressively stronger with depth, opposite the pattern observed with lawn soils (Table 3.2). Soil TN beneath nonturf cover exhibited no significant relationship with soil texture ($P < 0.05$), and only at depths of 0-10 cm was there a significant trend between nonturf soil TN and elevation ($r^2 = 0.17$, $P = 0.0209$). Soil texture, it is important to note, was typically measured for lawn soils only.

C:N ratio

Site-averaged soil TN and soil organic C were strongly related at all depths (Fig. 3.4). This correlation was also apparent in nearly constant average soil C:N ratio (organic C:total N) across the residential age gradient. A significant relationship between C:N ratio and yard age was observed only at the 10-20 cm depth and only for lawn soils ($r^2 = 0.17$, $P = 0.0164$). Average lawn soil C:N was significantly different from nonturf soil C:N at both 0-10 and 10-20 cm increments when one outlier was removed to preserve normality (Student's t-test, $P < 0.05$; Fig. 3.5). No significant difference between cover

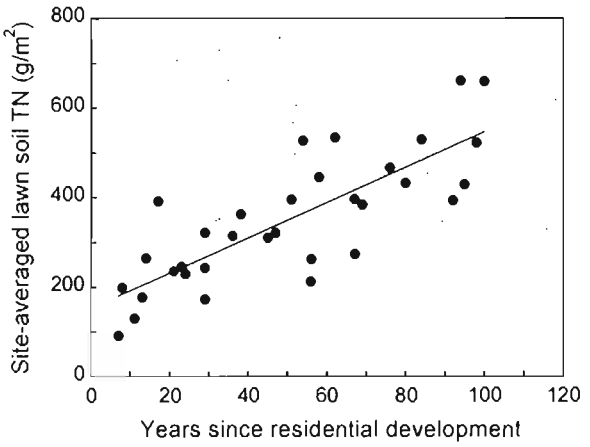

Fig. 3.3 Soil total nitrogen (TN) content of residential lawn surface soils (0-20 cm) as a function of yard age ($r^2 = 0.64$, $y = 3.9x + 154$). Each filled marker indicates the average value from replicate soil cores collected at a single site. TN was calculated individually for 0-10 and 10-20 cm depth increments of each replicate core, averaged by depth increment, and then summed. For 18 of the 38 sites, one or more core increment could not be included in the calculation due to obstructions encountered while coring.

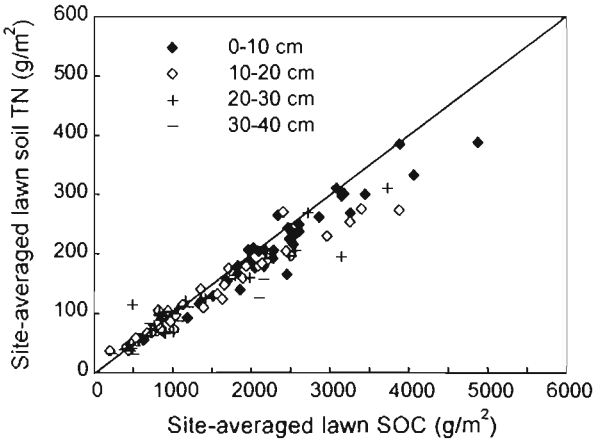

Fig. 3.4 Relationship between lawn soil total nitrogen (TN) and soil organic carbon (SOC) stocks by depth ($r^2 = 0.93$, $y = 0.08x + 14.2$).

types occurred below 20 cm.

Bulk soil $\delta^{15}N$

Lawn soil $\delta^{15}N$ measurements for the top 10 cm fell within the range of values reported by Kammerdiener (2004) for surface soils (0-2 cm) collected in unmanaged areas throughout the Salt Lake Valley (Fig. 3.6). Soil texture emerged as the site characteristic having greatest influence on bulk soil $\delta^{15}N$ (Table 3.3, Fig. 3.7). Soil $\delta^{15}N$ decreased as a function of percent sand and increased as a function of percent silt or percent clay. Regression models pairing soil $\delta^{15}N$ with percent sand or percent silt exhibited slopes significantly different from zero at all depths ($P < 0.05$). For nonturf soils, the relationship generally improved with depth, but for lawn soils the performance of these models remained nearly constant throughout the soil profile. There was a significant positive relationship between yard age and soil $\delta^{15}N$ only for lawn cover and only at 0-10 cm ($r^2 = 0.32$, $P = 0.0002$) and 10-20 cm ($r^2 = 0.29$, $P = 0.0011$). Moreover, when both yard age and percent silt were included in a multiple regression, the age parameter was significant only at 0-10 cm ($P < 0.05$). Elevation, which was correlated with soil texture at my sites, exhibited significant negative relationships with both turf and nonturf soil $\delta^{15}N$, and these relationships improved with soil depth. Average $\delta^{15}N$ of lawn soil was significantly different from nonturf soil $\delta^{15}N$ only at 0-10 cm.

Within-site variability

Variability in soil properties within residential yards was evaluated by comparing paired front and back lawn cores. At depths of 0-10 cm, the maximum difference in soil TN between paired cores was 164.3 g/m^2 and the standard deviation was 74.9 g/m^2. The

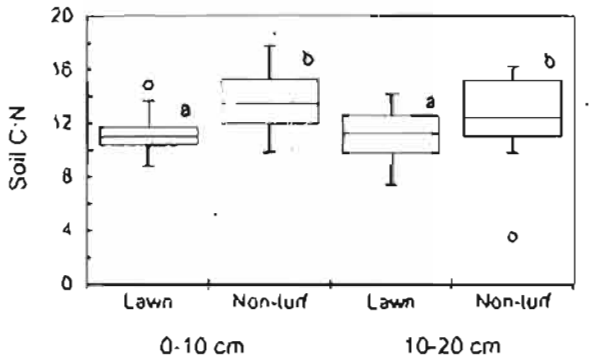

Fig 3.5 Ratio of percent organic carbon to percent total nitrogen in residential soils. Different letters indicate significant differences ($P < 0.05$). C:N beneath nonturf cover was significantly higher than that beneath lawn at both 0-10 and 10-20 cm.

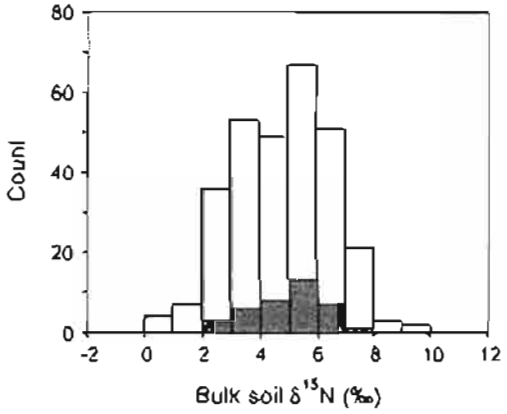

Fig. 3.6 Histogram of lawn (grey bars) and non-residential urban (open bars) bulk soil $\delta^{15}N$. Lawn values represent average $\delta^{15}N$ of replicate 0-10 cm core increments taken at each residential site. Nonresidential urban data come from Kammerdiener (2004). These soils were collected to 2 cm from a variety of unmanaged urban features. Values used here are the averages of two samples taken at different dates (March 2003 and April 2003) at each site.

Table 3.2 Site characteristics regressed against average soil total nitrogen (TN). R^2 is reported by soil depth (cm).

		Lawn soil TN				Nonturf soil TN			
Variable	Range	0-10	10-20	20-30	30-40	0-10	10-20	20-30	30-40
Age (yr)	7-100	0.53^{***}	0.64^{***}	0.29^{*}	0.20^{*}	0.13^{*}	0.19^{*}	0.36^{*}	0.72^{**}
Elevation (m)	1285-1525	0.31^{**}	0.26^{**}	0.30^{**}	0.28^{*}	0.17^{*}	0.15	0.05	0.08
%Sand	24-87	0.14^{*}	0.20^{**}	0.06	0.28^{*}	0.02	0.14	0.01	0.05
%Silt	10-61	0.20^{**}	0.31^{**}	0.12	0.36^{**}	0.03	0.17	0.05	0.12
%Clay	3-28	0.01	0.01	0.00	0.06	0.01	0.05	0.02	0.01
Age x elevation		0.66^{***}	0.70^{***}	0.46^{**}	0.36^{*}	0.23^{*}	0.31^{*}	0.39^{*}	0.78^{**}
Age x %Silt		0.55^{***}	0.67^{***}	0.32^{*}	0.41^{*}	0.13	0.26	0.36	0.72^{**}

$^{***}P < 0.0001$, $^{**}P < 0.01$, $^{*}P < 0.05$

Table 3.3 Site characteristics regressed against average lawn and nonturf bulk soil $\delta^{15}N$. R^2 is reported by soil depth (cm).

		Lawn soil $\delta^{15}N$				Nonturf soil $\delta^{15}N$			
Variable	Range	0-10	10-20	20-30	30-40	0-10	10-20	20-30	30-40
Age (yr)	7-100	0.32^{**}	0.29^{**}	0.00	0.01	0.11	0.10	0.19	0.24
Elevation (m)	1285-1525	0.18^{**}	0.31^{**}	0.39^{**}	0.52^{**}	0.25^{**}	0.14	0.32^{*}	0.46^{*}
%Sand	24-87	0.44^{***}	0.50^{***}	0.43^{**}	0.33^{*}	0.27^{**}	0.24^{*}	0.42^{**}	0.59^{**}
%Silt	10-61	0.50^{***}	0.59^{***}	0.44^{**}	0.37^{**}	0.29^{**}	0.29^{**}	0.51^{**}	0.71^{**}
%Clay	3-28	0.19^{**}	0.18^{*}	0.24^{*}	0.15	0.14^{*}	0.10	0.15	0.24
TN (g/m^2)		0.13^{*}	0.29^{**}	0.09	0.50^{**}	0.18^{*}	0.22^{*}	0.29^{*}	0.32
%Silt x Age		0.57^{***}	0.62^{***}	0.50^{**}	0.38^{*}	0.39^{**}	0.34^{*}	0.63^{**}	0.76^{**}
%Silt x TN		0.50^{***}	0.61^{***}	0.45^{**}	0.57^{**}	0.40^{**}	0.37^{*}	0.66^{**}	0.79^{**}

$^{***}P < 0.0001$, $^{**}P < 0.01$, $^{*}P < 0.05$

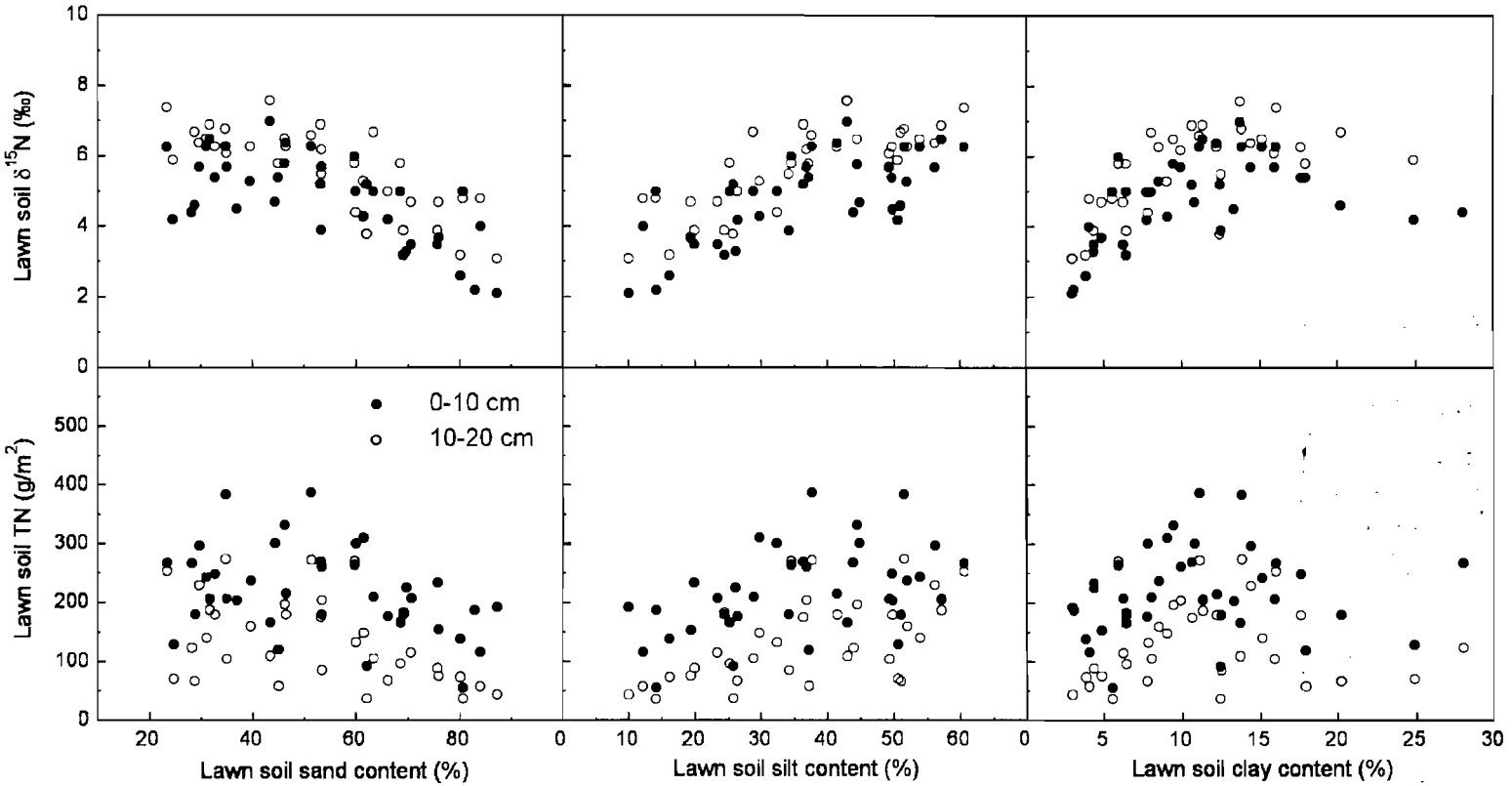

Fig 3.7 Lawn soil $\delta^{15}N$ and total nitrogen (TN) as a function of soil texture. Each marker indicates the average value from replicate cores (generally two) taken from a single site. Soil texture was generally measured on the upper 20 cm of lawn soil only.

same values at depths of 10-20 cm were 122.9 and 54.4 g/m^2, respectively. When TN of lawn surface soils was plotted as a function of yard age, there was no significant difference between the least squares means of front and rear yard soil TN (Student's *t*-test, $P < 0.05$). Soil $\delta^{15}N$ of back yard samples was significantly higher than soil $\delta^{15}N$ of front yards, but only at 0-10 cm (paired *t*-test, $P < 0.05$; Fig. 3.8). At this depth the maximum difference in soil $\delta^{15}N$ between front and back lawn cores was 2.5‰ and the standard deviation was 1.0‰. The same statistics at 10-20 cm were 4.2 and 1.6‰, respectively. Average lawn soil TN of residential sites decreased progressively down the soil profile (Fig. 3.9). Average $\delta^{15}N$ increased slightly, while C:N ratio remained nearly constant.

Discussion

Concerns about water quality have prompted considerable research into the transformation and movement of fertilizer N applied to turf (Petrovic et al. 1990). Studies using ^{15}N as a tracer have demonstrated the presence of 9 to 51% of labeled fertilizer N in turfgrass soils several months to several years after application (Petrovic et al. 1990; Miltner et al. 1996; Engelsjord et al. 2004; Frank et al. 2006; Raciti et al. 2008). The amounts recovered vary by study, as well as by fertilizer type, turfgrass species, clipping management, and rate and season of fertilizer application. Little work has been done to characterize N sequestration in lawn soils over longer time periods. Moreover, the extent to which climate influences soil N retention in urban ecosystems is not evident from past research. My results from an urban age gradient in the semiarid Salt Lake Valley, UT indicate that long-term accumulation of N in urban soils is likely, that changes in soil N

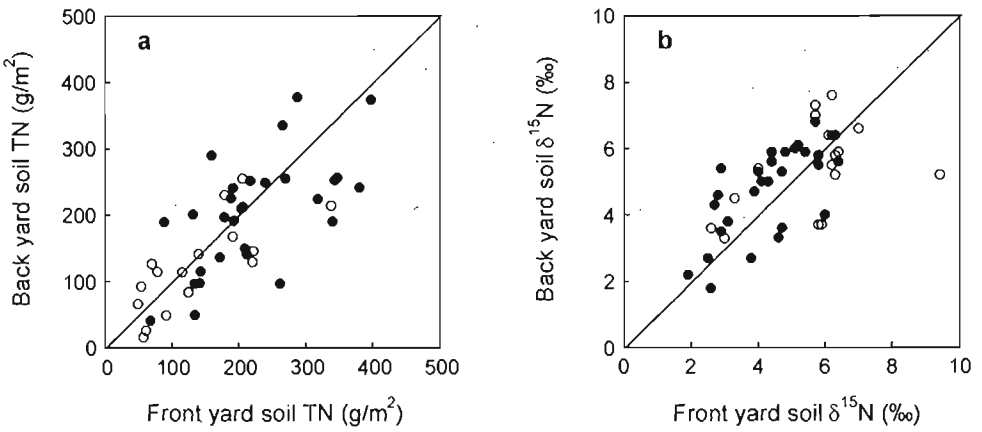

Fig 3.8 Variability in **a** soil total nitrogen (TN) and **b** bulk soil $\delta^{15}N$ within residential lawns, as measured by replicate lawn soil cores. One core each was generally collected from the front and back yard. Filled markers represent a depth of 0-10 cm. Open markers represent 10-20 cm. No significant difference was observed between TN of front and back cores at either depth (paired *t*-test, $P < 0.05$). Bulk soil $\delta^{15}N$ was significantly higher in back yards than front yards at 0-10 cm, but not at 10-20 cm ($P < 0.05$).

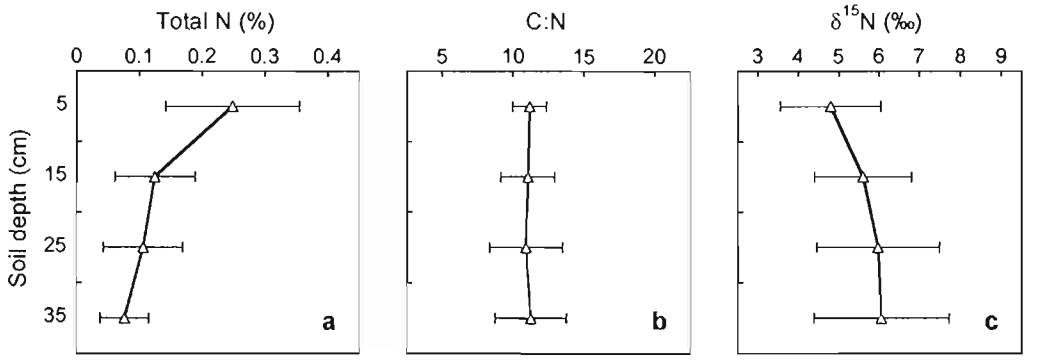

Fig. 3.9 Trends in lawn soil **a** total nitrogen, **b** C:N and **c** $\delta^{15}N$ with depth. Soil cores were collected and analyzed in 10 cm increments. Mean values of replicate cores from each site were averaged by depth. Error bars indicate one standard deviation from the mean.

stocks are tightly linked to changes in organic matter, and that gains in soil N may persist for more than a century in semiarid regions.

Among residential properties ranging in age from 7 to 100 years, a highly significant positive relationship was observed between yard age and soil TN stocks in the top 20 cm of lawns (Fig. 3.3). Average soil TN stocks to this depth were predicted to more than double following a century of residential land use, based on standard linear regression. Long-term accumulation of soil N has been reported in other US cities. Porter et al. (1980) compared soil organic N (SON) stocks of 100 turfgrass sites ranging in age from 1 to 125 years. Their study on Long Island, NY indicated a trend toward rapid increase in SON after lawn establishment, followed by saturation of SON gains after about 30 years. Lewis et al. (2006) estimated that soil N stocks in residential soils of the Phoenix, AZ metropolitan area grow at a rate of 1.2-1.9 g m^{-2} yr^{-1} in the surface 10 cm. The upper bound of their estimate mirrors the average N sequestration rate predicted here for the top 10 cm of residential lawn soils in the Salt Lake Valley. The steeper increase exhibited by my data is not surprising, considering that the Phoenix study did not differentiate between turf and nonturf cover. My measurements suggest that N does, on average, increase in nonturf areas, but that the predicted rate of increase (0.8 g m^{-2} yr^{-1}) is considerably lower than that observed for lawns.

It is well established that organic forms of N typically dominate the soil N pool (Post et al. 1985). I observed a tight linear relationship between TN and organic C in soil samples taken from all depths, beneath all ground covers, and across all sites (Fig. 3.4). Gains in TN thus appear to be directly proportional to gains in soil organic C beneath residential yards. Average C:N ratio of residential soils varied little as a function of yard

age, with significant trends occurring only at 10-20 cm beneath lawns. Litter and carbohydrate inputs from lawns in my study area evidently fuel robust activity by heterotrophic soil microbes, stimulating N mineralization, immobilization and incorporation into SOM. Kaye et al. (2005) found soil respiration rates from lawns in northern Colorado to be about three times greater than that from nearby native grassland, providing further evidence for heightened microbial activity in lawn soils. Consistent soil moisture from lawn irrigation was also discussed as a factor contributing to the sustained high soil respiration rates of lawns. My findings suggest that residential soils may retain a portion of applied N for at least as long as organic matter continues to accumulate. Yard age was the site characteristic explaining the greatest amount of intersite variation in soil TN to 20 cm (Table 3.2). Soil texture and elevation were of diminished importance in surface soils, but explained similar amounts of variability as yard age at deeper increments. It is in fact difficult to separate the effects of these three factors below 20 cm using my data, since correlations between soil texture and yard age, as well as between soil texture and elevation, were observed among residential sites. The same site characteristics identified here as important for predicting N stocks among urban sites were also found to be most successful in explaining intersite variation in soil organic C (Chapter 2).

The close association between soil C and N stocks, together with the characteristically low SOM content of most arid and semiarid ecosystems, hints at regional variation in the response of urban soils to supplemental N. In a similar study conducted in the northeastern US, Porter et al. (1980) predicted that average SON gains beneath turf were likely to plateau within about 30 years of turfgrass establishment.

Residential soils sampled in the Salt Lake Valley indicate the potential for N to accumulate in lawn surface soils for more than a century (Fig. 3.3). The discrepancy between these studies can be explained in part by regional differences in SOM content prior to conversion to residential use. Lawns in semiarid regions typically receive supplemental nutrients and irrigation to maintain productivity throughout the growing season. As a result, they are likely to exhibit comparable primary productivity and litter quality to lawns in wetter climates. All other factors being equal, the homogeneity of urban vegetation suggests that SOM content of residential yards should converge on similar end-points regardless of natural precipitation levels (Pouyat et al. 2003). Humid temperate ecosystems such as those common to the northeastern US typically have higher SOM pools than semiarid ecosystems (Jobbágy and Jackson 2000). They are therefore likely to reach steady state sooner and have lower total SON gains following transition to lawn cover. Such regional differences in SON gains might be less striking for lawns established on farmland, since agricultural practices also tend to dampen geographic differences in native SOM content (Post and Mann 1990).

A second possibility arises from study design. Older neighborhoods in the Salt Lake Valley are primarily clustered in the northeast corner of the valley (Fig 3.1). This trend, together with the relatively small population of old homes, resulted in study sites older than 60 years having a smaller geographic distribution than younger sites. If pre-urban soil TN stocks of the oldest parcels were systematically higher than at other properties, the regression between yard age and TN would overestimate both the magnitude and duration of N accumulation. Such a scenario might conceal the fact that some sites had already reached their maximum capacity to store applied N. Yard age was found to be

much more effective at explaining variation in lawn surface soil TN than was elevation or soil texture, a point that appears to diminish the importance of parent material effects. It is nonetheless conceivable that site characteristics such as topography, hydrology, or historic land use promoted systematically high soil TN stocks in certain portions of the valley prior to urban development. This possibility is difficult to test directly, since areas retaining their original condition are rare or absent from the oldest portions of the Salt Lake Valley metropolitan area.

N isotope natural abundance has been identified as a potential tool for monitoring and predicting ecosystem response to increased N availability (Amundson et al. 2003, Pardo et al. 2006). A small body of research indicates that N amendments can influence soil and plant N isotope ratios in predictable ways. Högberg (1990) observed elevated $^{15/14}N$ in the understory vegetation of forest sites receiving 18 years of experimental fertilizer applications. A positive correlation was subsequently documented between pine needle ^{15}N abundance and fractional losses of fertilizer N (Högberg and Johannisson 1993). Similar results were reported for an uncontrolled setting in the San Bernardino Mountains, CA, where $^{15/14}N$ of bulk soil and conifer needles was significantly higher at sites receiving greater N deposition (Korontzi et al. 2000).

It is useful to distinguish here between N saturation, typically defined as the point at which N inputs exceed plant and microbial demand, and the point at which added N ceases to accumulate in plants and soils. Little experimental work has been done to characterize changes in N stable isotope ratios as fertilized soils return to steady state. A mass-balance model proposed by Brenner et al. (2001) indicates that the N isotope ratio of bulk soil at steady state is contingent upon just two factors: the isotope ratio of inputs

to the plant-soil system (assumed to be nearly constant among most natural settings) and the apparent fractionation of N outputs. If such constraints hold true for fertilized ecosystems, any changes in bulk soil N isotope ratio brought about by fertilization are likely to stabilize as soil N dynamics approach a new equilibrium. Across the urban age gradient sampled in this study, site-averaged lawn soil $\delta^{15}N$ increased significantly with yard age, but only at depths of 0-10 cm and 10-20 cm (Table 3.3). Bulk soil $\delta^{15}N$ at 0-10 cm was also significantly higher for lawn than nonturf cover types. This trend toward enrichment of lawn soil N over time appears consistent with increased nitrification, denitrification, NH_3 volatilization and leaching likely to occur from fertilization and watering. No strong conclusions can be made regarding the shape of this trend, however. As a result, the N isotope data presented here likely provide little clue as to whether or not sites have saturated in their capacity to sequester N. Interpretations are further complicated by the possibility that the isotopic composition of N inputs and outputs has changed over time. Animal-derived fertilizers, for example, were likely much more prevalent prior to mass production of N fertilizers using the Haber-Bosch process and are on average considerably more enriched in ^{15}N (Vitoria 2004).

At most, yard age appears to be a secondary factor influencing the N isotope composition of residential soils in the Salt Lake Valley. Soil texture was the site characteristic most effective at explaining variation in bulk soil $\delta^{15}N$ among lawns (Table 3.3). Soil $\delta^{15}N$ was negatively correlated with percent sand at all depths and positively correlated with percent silt and percent clay. The relationship between $\delta^{15}N$ and silt was clearly linear, but relationships involving sand or clay were best modeled using polynomial trend lines. A correlation between soil N isotope ratio and texture has been

observed in unmanaged and agricultural soils as well (Shearer et al. 1978; Sutherland et al. 1993) and likely arises from texture-specific differences in the proportion of N lost through fractionating vs. non-fractionating pathways. Clay and silt-sized soil particles are known to protect and stabilize SOM through chemical and physical associations (Hassink 1997; Mikutta et al. 2006). Soils high in clay and silt are thus expected to retain a greater amount of organic matter, which is likely to become enriched in ^{15}N through humification and the loss of depleted N by nitrification and denitrification (Natelhoffer and Fry 1988; Kramer et al. 2003). Nonfractionating losses of dissolved organic N via leaching, meanwhile, are likely to be more prevalent in coarse-textured soils. This explanation is supported by the significant, albeit weak, relationship observed here between soil texture and soil TN content of residential lawns. It is also consistent with a significant relationship between TN and $\delta^{15}N$. Regardless of the cause, it is evident that native soil properties of this urban ecosystem more strongly influence bulk soil $\delta^{15}N$ than do the continual input of fertilizer and the accumulation of N. What is not clear from this work is whether or not lawn management amplifies the relationship between $\delta^{15}N$ and soil texture compared to natural settings. Nevertheless, interpretations of N stable isotope natural abundance should bear in mind the potentially large variability associated with soil texture.

Conclusions

Total N appears to increase over time beneath lawn and other pervious surfaces of residential yards in the Salt Lake Valley. Accumulation in the upper 20 cm of lawn soil may occur at an average rate of about 3.9 g N $m^2 yr^{-1}$ and may occur for more than a century, based on linear regression pairing soil TN of residential lawns to parcel age. In

comparison, recommended lawn fertilizer application rates in the US have been reported to be about 15 g N m^2 yr^{-1} (Hall et al. 2008). Measurements of lawn clipping N indicate that 11-12 g N m^2 yr^{-1} is lost from the turfgrass system if clippings are disposed off-site (Kaye et al. 2005, Golubiewski 2006). Soil retention of applied N may thus figure prominently into the N cycle of residential yards and contribute to the tight cycling of N often reported for turfgrass systems (e.g., Raciti et al. 2008). Increases in SOM over time appear to facilitate N retention in residential soils of the Salt Lake Valley. My results suggest that the same site characteristics influencing the size of SOC stocks in my study area also influence the size of soil TN stocks. The relatively low native SOM content of most arid and semiarid ecosystems may result in greater total gains in soil N following lawn establishment than observed in wetter climates.

It is apparent from this and previous studies that urban landscaping practices have the potential to considerably modify regional soil N stocks. As the need to conserve water in arid and semiarid regions leads homeowners to replace existing lawns with vegetation requiring less water, the fate of inflated N stocks beneath old lawns remains uncertain. Removal of forest and other vegetation has been shown to promote the loss of soil N in some cases (Schesinger 1991). Such conditions may enhance mineralization and nitrification by increasing soil temperatures, and they may temporarily promote N leaching, volatilization and runoff by reducing plant and mycorrhizal N uptake (Aber et al. 1998). Similar processes might be expected when old lawns are removed and subsequently replaced with sparser and less productive vegetation (e.g., xeriscaping). Understanding the fate of N applied to lawns in semiarid regions may ultimately require

more research into the effects of xeriscaping and similar landscaping practices on soil N dynamics.

References

Aber J, McDowell W, Nadelhoffer K, Magill A, Berntson G, Kamakea M, McNulty S, Currie W, Rustad L, Fernandez I (1998) Nitrogen saturation in temperate forest ecosystems BioScience 48:921-934

Baker LA, Hope D, Xu Y, Edmonds J, Lauver L (2001) Nitrogen balance for the Central Arizona-Phoenix (CAP) ecosystem. Ecosystems 4:582-602

Bijoor NS, Czimczik CI, Pataki DE, Billings SA (2008) Effects of temperature and fertilization on nitrogen cycling and community composition of an urban lawn. Global Change Biology 14:1-13

Brenner DL, Amundson R, Baisden WT, Kendall C, Harden J (2001) Soil N and ^{15}N variation with time in a California annual grassland ecosystem. Geochimica et Cosmochimica Acta 65:4171-4186

Delgado J A, Mosier AR, Valentine DW, Schimel DS, Parton WJ (1996) Long term ^{15}N studies in a catena of the shortgrass steppe. Biogeochemistry 32:41-52

Easton ZM, Petrovic AM (2004) Fertilizer source effect on ground and surface water quality in drainage from turfgrass. Journal of Environmental Quality 33:645-655

Elser JJ, Marzolf ER, Goldman CR (1990) Phosphorus and nitrogen limitation of phytoplankton growth in the freshwaters of North America: a review and critique of experimental enrichments. Canadian Journal of Fisheries and Aquatic Sciences 47:1468-1477

Engelsjord ME, Branham BE, Horgan BP (2004) The fate of nitrogen-15 ammonium sulfate applied to Kentucky bluegrass and perennial ryegrass turfs. Crop Science 44:1341-1347

Fenn ME, Haeuber R, Tonnesen GS, Baron JS, Grossman-Clarke S, Hope D, Jaffe DA, Copeland S, Geiser L, Rueth HM, Sickman JO (2003) Nitrogen emissions, deposition, and monitoring in the western United States. BioScience 53:391-403

Fenn ME, Poth MA, Aber JD, Baron JS, Bormann BT, Johnson DW, Lemly D, McNulty SG, Ryan DF, Stottlemyer R (1998) Nitrogen excess in North American ecosystems: predisposing factors, ecosystem responses, and management strategies. Ecological Applications 8:706-733

Frank KW, O'Reilly KM, Crum JR, Calhoun RN (2006) The fate of nitrogen applied to a mature Kentucky bluegrass turf. Crop Science 46:209-215

Golubiewski NE (2006) Urbanization transforms prairie carbon pools: effects of landscaping in Colorado's Front Range. Ecological Applications 16:555-571

Goodale CL, Aber JD (2001) The long-term effects of land-use history on nitrogen cycling in northern hardwood forests. Ecological Applications 11:253-267

Guillard K, Kopp KL (2004) Nitrogen fertilizer form and associated nitrate leaching from cool-season lawn turf. Journal of Environmental Quality 33:1822-1827

Hall SJ, Huber D, Grimm NB (2008) Soil N_2O and NO emissions from an arid, urban ecosystem. Journal of Geophysical Research 113: G01016, doi:10.1029/2007JG000523

Hassink J (1997) The capacity of soils to preserve organic C and N by their association wilt clay and silt particles. Plant and Soil 191:77-87

Hobbs F, Stoops N (2002) U.S. Census Bureau, Census 2000 Special Reports, Series CENSR-4, Demographic Trends in the 20th Century, U.S. Government Printing Office, Washington, DC

Högberg P (1990) Forests losing large quantities of nitrogen have elevated 15N:14N ratios. Oecologia 84:229-231

Högberg P (1997) ^{15}N natural abundance in soil-plant systems. New Phytologist 137:179-203

Högberg P, Johannison C (1993) 15N abundance of forests is correlated with losses of nitrogen. Plant Soil 157:147-150

Högberg P, Tamm CO, Högberg M (1992) Variations in 15N abundance in a forest fertilization trial: Critical loads of N, N saturation, contamination and effects of revitalization fertilization. Plant Soil 142:211-219

Homer C, Dewitz J, Fry J, Coan M, Hossain N, Larson C, Herold N, McKerrow A, VanDriel JN, Wickham J (2007) Completion of the 2001 National Land Cover Database for the conterminous United States. Photogrammetric Engineering and Remote Sensing 73:337-341

Horgan BP, Branham BE, Mulvaney RL (2002) Direct measurement of denitrification using ^{15}N-labeled fertilizer applied to turfgrass. Crop Science 42:1602-1610

Howarth RW (1988) Nutrient limitation of net primary production in marine ecosystems. Annual Review of Ecology and Systematics 19:89-110

Howarth RW, Boyer EW, Pabich WJ, Galloway JN (2002) Nitrogen use in the United States from 1961-2000 and potential future trends. Ambio 31:88-96

Hutchinson GL, Mosier AR (1979) Nitrous oxide emissions from an irrigated cornfield. Science 205:1125–1127

Jo H, McPherson EG (1995) Carbon storage and flux in urban residential greenspace. Journal of Environmental Management 45:109-133

Jobággy EG, Jackson RB (2000) The vertical distribution of soil organic carbon and its relation to climate and vegetation. Ecological Applications 10:423-436

Johnson DW, Cheng W, Burke IC (2000) Biotic and abiotic retention in a variety of forest soils. Soil Science Society of America Journal 64:1503-1515

Kammerdiener SA (2004) Isotopes in urban cheatgrass quantify atmospheric pollution. MS Thesis, University of Utah, 104 pp

Kaye JP, Burke IC, Mosier AR, Guerschman JP (2004) Methane and nitrous oxide fluxes from urban soils to the atmosphere. Ecological Applications 14:975-981

Kaye JP, McCulley RL, Burke IC (2005) Carbon fluxes, nitrogen cycling, and soil microbial communities in adjacent urban, native and agricultural ecosystems. Global Change Biology 11:575-587

Kettler TA, Doran JW, Gilbert TL (2008) Simplified method for soil particle size determination to accompany soil-quality analyses. Soil Science Society of America Journal 65:849-852

Korontzi S, Macko SA, Anderson IC, Poth MA (2000) A stable isotopic study to determine carbon and nitrogen cycling in a disturbed southern Californian forest ecosystem. Global Biogeochemical Cycles 14:177-188

Kramer MG, Sollins P, Sletten RS, Swart PK (2003) N isotope fractionation and measures of organic matter alteration during decomposition. Ecology 84:2021-2025

Lewis DB, Kaye JP, Gries C, Kinzig AP, Redman CL (2006) Agrarian legacy in soil nutrient pools of urbanizing arid lands. Global Change Biology 12:703-709

Mikutta R, Kleber M, Torn MS, Jahn R (2006) Stabilization of soil organic matter: association with minerals or chemical recalcitrance? Biogeochemistry 77:25-56

Miltner ED, Branham BE, Paul EA, Rieke PE (1996) Leaching and mass balance of ^{15}N-labeled urea applied to a Kentucky bluegrass turf. Crop Science 36:1427-1433

Mosier A, Schimel D, Valentine D, Bronson K, Parton W (1991) Methane and nitrous oxide fluxes in native, fertilized and cultivated grasslands. Nature 350:330-332

Nadelhoffer KJ, Downs MR, Fry B (1999) Sinks for N-15 enriched additions to an oak forest and a red pine plantation. Ecological Applications 9:72-86

Nadelhoffer KJ, Fry B (1988) Controls on natural nitrogen-15 and carbon-13 abundances in forest soil organic matter. Soil Science Society of America Journal 52:1633-1640

Pardo L et al. (2006) Regional assessment of N saturation using foliar and root $\delta^{15}N$. Biogeochemistry 80:143-171

Pardo LH, Hemond HF, Montoya JP, Fahey TJ, Siccama TG (2002) Response of the natural abundance of 15N in forest soils and foliage to high nitrate loss following clear-cutting. Canadian Journal of Forest Research 32:1126-1113

Petrovic AM (1990) The fate of nitrogenous fertilizers applied to turfgrass. Journal of Environmental Quality 19:1-14

Porter KS, Bouldin DR, Pacenka S, Kossack RS, Shoemaker CA, Pucci AA Jr (1980) Studies to assess the fate of nitrogen applied to turf: Part 1. Research Project Technical Completion Report OWRT Project A-086-NY. Cornell University

Post WM, Mann LK (1990) Changes in soil organic carbon and nitrogen as a result of cultivation. In Bouwman AF (ed) Soils and the greenhouse effect. John Wiley and Sons Ltd., West Sussex, England, pp 401-406

Post WM, Pastor J, Zinke PJ, Stangenberger AG (1985) Global patterns of soil nitrogen storage. Nature 317:613-616

Pouyat RV, Russell-Anelli J, Yesilonis ID, Groffman PM (2003) Soil carbon in urban forest ecosystems. In J.M. Kimble et al. (ed.) The potential of U.S. forest soils to sequester carbon and mitigate the greenhouse effect. CRC Press, Boca Raton, FL, pp 347-362

Qian Y, Follet RF (2002) Assessing soil carbon sequestration in turfgrass systems using long-term soil testing data. Agronomy Journal 94:930-935

Raciti SM, Groffman PM, Fahey TJ (2008) Nitrogen retention in urban lawns and forest. Ecological Applications 18:1615-1626

Robinson D (2001) Delta-15 N as an integrator of the nitrogen cycle. Trends in Ecology and Evolution 16:153-162

Schlesinger WH (1991) Biogeochemistry: an analysis of global change. Academic Press, Inc., San Diego

Shearer G, Kohl DH, Chien SH (1978) The nitrogen-15 abundance in a wide variety of soils. Soil Science Society of America Journal. 42:899-902

Sherrod LA, Dunn G, Peterson GA, Kolberg RL (2002) Inorganic carbon analysis by modified pressure-calcimeter method. Soil Science Society of America Journal 66:299-305

Shi W, Dell E, Bowman D, Iyyemperumal K (2006) Soil enzyme activities and organic matter composition in a turfgrass chronosequence. Plant Soil 288:285-296

Star JL, DeRoo HC (1981) The fate of nitrogen applied to turfgrass. Crop Science 21:531-536

Sutherland RA, van Kessel C, Farrell RE, Pennock DJ (1993) Landscape-scale variations in plant and soil nitrogen-15 natural abundance. Soil Science Society of America Journal 57:169-178

Tan KH (1996) Soil sampling, preparation and analysis. Marcel Dekker, Inc., New York

Vitoria L, Otero N, Soler A, Canals A (2004) Fertilizer characterization: isotopic data (N, S, O, C, and Sr). Environmental Science and Technology 38:3254-3262

Vitousek PM, Howarth RW (1991) Nitrogen limitation on the land and in the sea: how can it occur? Biogeochemistry 13:87-115

Vitousek PM, Aber JD, Howarth RW, Likens GE, Matson PA, Schindler DW, Schlesinger WH, Tilman DG (1997) Human alteration of the global nitrogen cycle: source and consequences. Ecological Applications 7:737-750

Wagner SW, Hanson JD, Olness A, Voorhees WB (1998) A volumetric inorganic carbon analysis system. Soil Science Society of America Journal 62:690-693

Wakefield JH (1933) A study of the plant ecology of the Salt Lake and Utah Valleys before the Mormon immigration. MS thesis, Brigham Young University, 56 pp

WorldClimate (2008) http://www.worldclimate.com

Zhu W, Hope D, Gries C, Grimm NB (2006) Soil characteristics and the accumulation of inorganic nitrogen in an arid urban ecosystem. Ecosystems 9:711-724

CHAPTER 4

ABOVEGROUND TREE CARBON AND NITROGEN STOCKS ALONG AN URBAN AGE GRADIENT IN SEMIARID SALT LAKE VALLEY, UT

Introduction

Only about 25% of the Intermountain West, USA is forested (USDA Forest Service 2008a). The largest cities in the Intermountain West occupy areas originally characterized by grassland, shrub steppe or desert. Although the types of vegetation surrounding these cities are diverse, the prevalent use of trees and turfgrass in urban landscaping has resulted in many homologous features within the urban cores. Urban forests have expanded across many treeless landscapes of the western US, resulting in a fundamental shift in the ratio of aboveground to belowground biomass (Golubiewski 2006).

In this chapter I compare measurements of aboveground tree carbon and nitrogen stocks in the Salt Lake Valley urban ecosystem to measurements of soil C and N stocks reported in earlier chapters. Similar to urban areas throughout the US, the Salt Lake Valley metropolitan area is characterized by extensive cover of urban forest. According to the 2001 National Land Cover Dataset, average tree cover in urbanized portions of the Salt Lake Valley was 16% in 2001 (Homer et al. 2007; Fig. 4.1). Most historical descriptions of the Salt Lake Valley prior to or immediately following European settlement suggest that tree cover in the region was initially close to zero (Wakefield

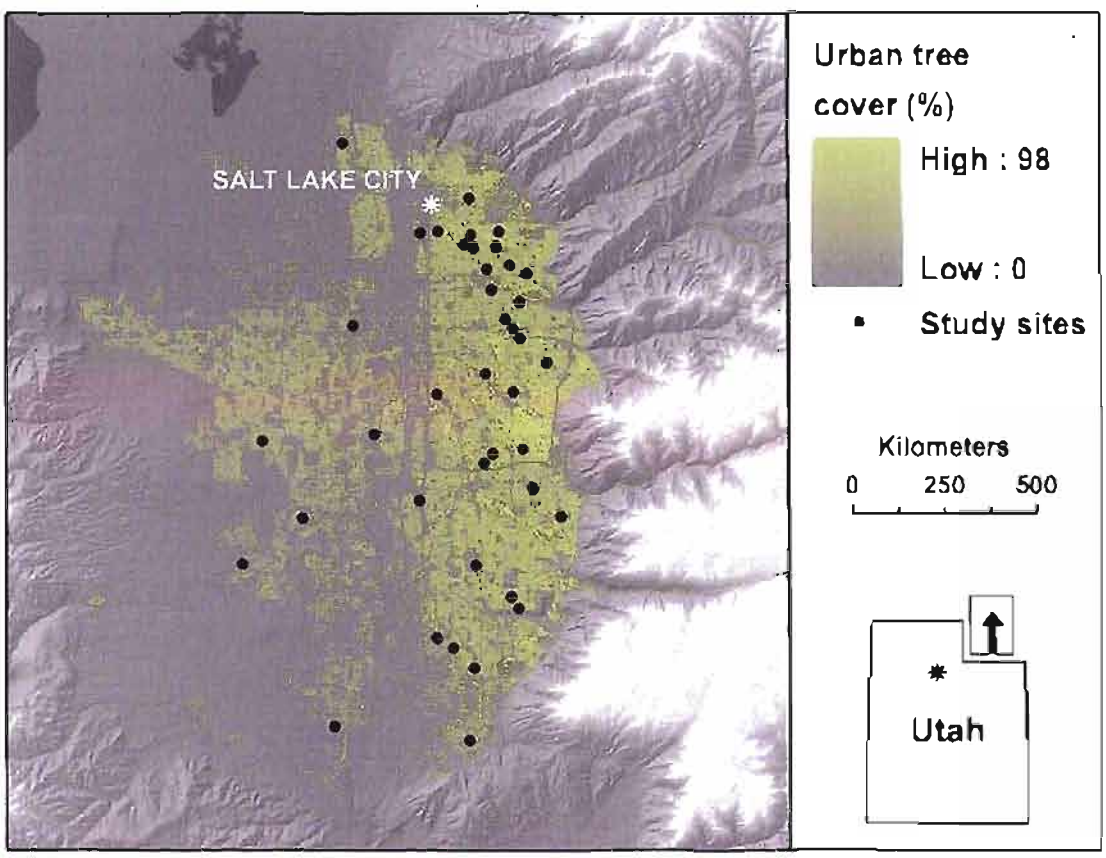

Fig. 4.1 Urban tree cover in the Salt Lake Valley derived from the 2001 National Land Cover Dataset (Homer et al. 2007). Urban tree cover was estimated by clipping the original 2001 NLCD tree cover dataset to only those pixels classified as urban land cover.

1933). Information on the biomass, species distribution, structure and health of these urban forests is currently available only for street trees maintained by city governments and only for a portion of the Salt Lake Valley. No data have been systematically collected for trees growing on private property.

Trees are sinks for atmospheric C, storing a portion of the C assimilated during photosynthesis as wood and roots. Considering that trees were historically sparse or absent from developed regions such as the Salt Lake Valley, the cumulative contribution of urban trees to removal and storage of atmospheric C is potentially large. Concerns about rising levels of atmospheric CO_2 have spurred interest in strategies for sequestering C. Forest inventories in other US cities suggest that while urban forests can be net sinks of atmospheric C, they likely do little to offset the C emissions of urban residents (Nowak 1994; Nowak and Crane 2002). Such conclusions indicate only that anthropogenic C emissions in the US are extraordinarily large; they do not diminish the possibility that afforestation and other vegetation changes associated with urban development may dramatically alter the biogeochemical cycles of arid and semiarid ecosystems. Here, I summarize trends in biomass, species distribution and forest structure observed on residential sites in the Salt Lake Valley metropolitan area. This chapter is meant as both a complement to earlier sections of this thesis and an additional contribution to the evolving ecological history of the Salt Lake Valley and semiarid US.

Materials and methods

Study design

Trees were measured on 40 residential parcels located throughout the Salt Lake Valley, UT metropolitan area (Fig. 4.1). Concurrent measurements of soil C and N stocks

were made at 38 of these sites, as discussed in earlier chapters. Readers are referred to these chapters for more detailed descriptions about study site selection and characteristics. Briefly, study sites consisted of single-family residential parcels selected randomly from a Salt Lake County parcel database. Residential parcels were designated as the unit of study in order to assess the influence of site age on tree biomass and other urban forest characteristics. Selection of potential sites was stratified by decade of development (1900-2007), with the year of house construction substituted for the actual date a parcel was converted to residential use. Selection of parcels developed after 1979 was also stratified by prior land use (irrigated agriculture, dryland agriculture or range/wildland). Permission to access selected parcels was received for 40 sites. The oldest site was 100 years old and the youngest was 7. Parcel boundaries defined the extent of study plots, resulting in variable plot sizes ranging from 0.037 to 0.243 hectares, with a study average of 0.086 hectares. The maximum distance between sites was 33 km and the elevation range was 240 m. No effort was made to control for other potentially important factors such as neighborhood socioeconomic status or housing density.

Tree measurements

Sites were visited between May and November of 2007. All trees rooted within the parcel boundaries and larger than 2.4 cm diameter-at-breast-height (DBH) were tallied and measured. Tree locations were sketched on field maps created from 0.3 m resolution aerial photography and later digitized as point features in a geographic information system (GIS). This study did not include street trees rooted in the public "park-strip" between the parcel boundaries and the street. For each tree, DBH (1.37 m above ground level) was measured to the nearest millimeter using a diameter tape. Total height was

measured to the nearest decimeter using a laser rangefinder/inclinometer. Additional measurements included height from ground level to the lowest live foliage and two measurements of crown width: one in the north-south direction, bisecting the tree stem, and one in the east-west direction. Crown width was estimated when canopies extended onto neighboring parcels. In cases where wind damage, trimming, tree crowding, or other factors substantially reduced tree biomass, percent of missing crown volume was visually estimated. Missing crown volume was only noted when it exceeded 30% of an ideal, full crown.

Tree biomass

Regression equations relating DBH to total aboveground dry biomass were extracted from the literature. These equations were applied to my tree dataset using a script developed in a GIS. Dry aboveground biomass of each tree was calculated using biomass equations compiled for North American (Ter-Mikaelian and Korzukhin 1997; Jenkins et al. 2004) and European (Zianis et al. 2005) tree species (Appendix B). Together, these compilations include several to zero equations for each species encountered on my urban plots. When multiple equations were available for a single species, I followed suggestions presented in Ter-Mikaelian and Korzukhin (1997). Only equations developed near the study region were used when available. Otherwise, multiple equations for a single species were grouped into similar DBH size classes and their results were averaged. Equations were not included if developed from an unusually small sample size or if they lacked adequate metadata concerning site characteristics, number of trees used, and diameter range sampled. Because many urban trees in the Salt Lake Valley are not native to the region or are rarely dominant in a natural setting, few

equations specific to the study region were available. For the majority of species encountered in this study, equations were not available for any region. In such instances, equations established for a species in the same genus were substituted if that species exhibited an analogous growth habit. Otherwise, I applied general equations developed by Jenkins et al. (2003) for forest C accounting in the US. These authors provided individual equations for four hardwood groups, five softwood groups, and one woodland group. A list of US tree species belonging to each group was also provided. In cases where a species encountered on my urban sites was not represented in the list, I assigned it to a group containing the same (or most similar) genus, family, or growth habit. Equations from Jenkins et al. (2003) were also used to check for abnormal results obtained from species-specific biomass equations, when such equations were available.

All biomass equations used in this study were developed from trees in non-urban forest settings. Urban trees are generally more widely spaced and less shaded than their wild counterparts, a trait that may lead to fuller crowns and larger DBH to height ratios in urban trees (Nowak 1994). In addition, trimming often reduces the biomass of urban trees, and horticultural varieties may be bred for fuller crowns and shorter statures than native genotypes. Despite these differences, few biomass equations have been developed specifically for urban trees. Those that are available in the literature were derived using a limited number of species and are thus not broadly applicable. Preliminary findings by Nowak (1994) indicate that equations derived for natural settings likely overestimate the biomass of urban trees by about 20%. I followed the approach of Nowak (1994) by multiplying the calculated biomass of each tree in my study by a factor of 0.8. Biomass estimates were also reduced for observed crown losses greater than 30%. Loss-adjusted

biomass ($B_{adjusted}$) was calculated using the following equations:

$$B_{adjusted} = B_{calculated} - [B_{calculated} * (\% \text{ crown missing}/100) * \text{crown ht/tree ht}] \quad (4.1)$$

for trees whose height was measured, or

$$B_{adjusted} = B_{calculated} - [B_{calculated} * (\% \text{ crown missing}/100) * 0.8] \quad (4.2)$$

for trees lacking a height measurement. A factor of 0.8 was used in Equation 4.2 to approximate the average proportion, in my study, of total tree height occupied by live crown.

DBH correction for forked trees

Urban trees often exhibit marked branching below breast height. With regard to DBH measurements, I generally followed protocols established by the US Forest Service Forest Inventory and Analysis (FIA) program. My methods deviated in cases where severe forking occurred below breast height and above 30 cm. If a primary stem was no longer identifiable above such branching, DBH was measured on the main stem below the fork, rather than on each individual branch. This method likely overestimated stem diameter but greatly increased the efficiency and simplicity of diameter measurements of urban trees.

For trees whose stem diameter was measured either below or above breast height, DBH was estimated prior to biomass calculation and other analyses. This step was taken to reduce biomass overestimates and allow more meaningful comparisons among tree size classes. Regression equations were developed to describe trends in stem diameter as a function of measurement height. Twenty hardwood trees and 20 softwood trees were

sampled on the University of Utah campus in December 2007. Selection was not random. Trees were required only to be single-stemmed and have minimal branching below breast height, and a variety of species were sampled. DBH of selected trees ranged from 4.3 to 49.0 cm. Diameters at six heights (40, 60, 80, 100, 120, 137, and 160 cm) were recorded for each tree. Logarithmic trend lines were fitted separately to hardwood and softwood data, using height from ground level as the independent variable and the ratio of diameter to DBH as the dependent variable (Fig. 4.2). Regression models performed reasonably well for both tree classes ($r^2 = 0.70$ in both cases) and were subsequently used to estimate DBH of trees whose diameters were measured at alternative heights. A separate equation was used for either hardwood or softwood trees. DBH of woodland trees whose diameters were measured at root collar were estimated using a generalized equation developed by (Chognacky and Rogers 1999) from pinyon pine (*Pinus edulis*), Utah juniper (*Juniperus*

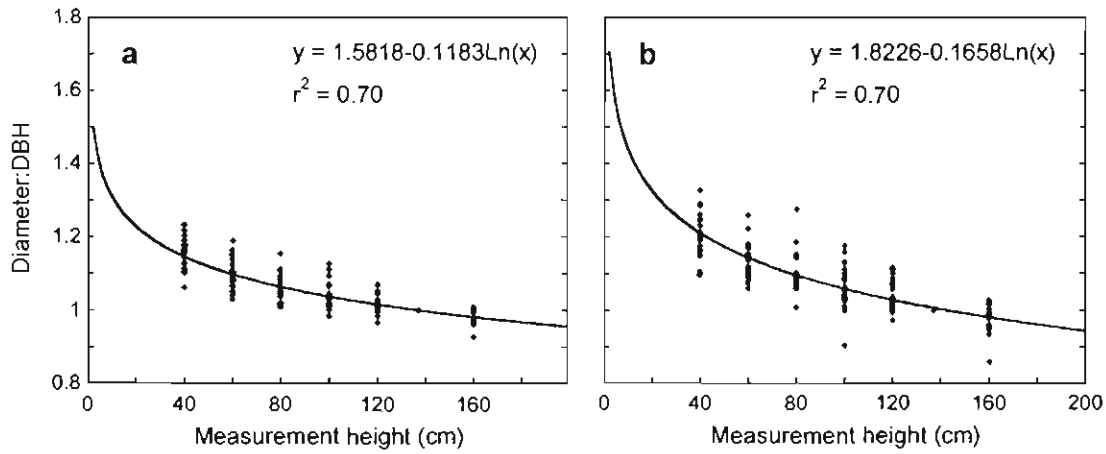

Fig. 4.2 Regression equations used to estimate tree DBH from measurements made above or below breast height. Separate equations were developed for **a** hardwood and **b** softwood species using 40 trees sampled on the University of Utah campus.

osteosperma) and Gambel oak (*Quercus gambelii*). Trees with estimated DBH smaller than 2.5 cm were not included in subsequent analyses.

Tree canopy cover estimates

Tree canopy cover was estimated for 38 of the 40 sites. Two sites were not included in this analysis due to incomplete information for tree height, crown height and crown width. Canopy cover estimates were calculated in a GIS using crown width measurements and tree locations recorded in the field. Two perpendicular crown diameter measurements – one in the north-south direction and one taken east-west – were averaged for each tree. Circular buffers were created in the GIS around each tree point feature. The width of the buffer was equal to the average crown radius of the corresponding tree. The resulting circular crown polygons were first merged and then dissolved to eliminate overlap among crowns. This procedure resulted in a single tree canopy polygon for each site. The GIS provided the area of each canopy polygon, and these values were divided by total parcel area to report the proportion of tree cover on each study parcel. Portions of tree crowns extending beyond parcel boundaries were included in the calculation. I assumed that, on average, the amount of crown cover arising from neighboring trees, which was not accounted for in my sampling, was similar to the area of canopy polygons extending onto adjacent lots. This approach also helped correct for street tree canopies that may have extended onto study parcels but were not sampled by my design.

Site summaries

Because study sites varied in size, I converted total aboveground tree biomass estimates to a per ground area basis. This adjusted estimate was calculated by summing

the biomass of all trees greater than 2.4 cm DBH and dividing by the parcel area, as reported by the Salt Lake County Assessor. Carbon and N stocks were simply scaled from these biomass estimates. Aboveground tree C content was assumed to be 50% of total dry biomass (Nowak and Crane 2002), and nitrogen stocks were calculated using a vegetation biomass C:N ratio of 160 (Schlesinger 1991). Tree basal area was calculated by summing the cross-sectional area at breast height of all trees on a parcel and dividing by parcel area. Homes, driveways and other impervious surfaces covered about 31% of each parcel, on average (Table 2.4). Since tree establishment is not possible in these areas, I also reported tree biomass, C, N and basal area per square meter of pervious surfaces. This was done for each parcel by dividing the measurement of interest by the total area covered by pervious surfaces.

Results and discussion

C and N storage in aboveground tree biomass

Afforestation of grassland, shrub steppe and farmland may be the most obvious impact of urbanization on ecosystem C and N storage in many semiarid regions. It is likely not, however, the process of greatest consequence. My results suggest that the largest changes in C and N stocks of residential parcels in the Salt Lake Valley typically occur below ground. Average aboveground tree C on residential sites was estimated to be 1458 g/m^2 for entire parcels and 2033 g/m^2 when calculated using only pervious surfaces (Table 4.1). If tree biomass prior to urbanization is assumed to have been near zero, these values represent reasonable estimates of net changes in aboveground tree C following residential development, since tree C on my study sites varied little as a function of yard age (Fig. 4.3). In comparison, average lawn soil organic carbon (SOC) in the upper 40 cm

Table 4.1 Average live and dead aboveground tree biomass, carbon storage, nitrogen storage, density and basal area measured on residential parcels (n = 40)

	Biomass (g/m^2)		Carbon (g/m^2)		Nitrogen (g/m^2)		Density (trees/ha)		Basal area (m/ha)	
	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE
Entire parcel										
Live	2884	574	1442	287	9.0	1.8	168	25	6.4	1.0
Dead	33	26	16	13	0.1	0.1	4	1	0.1	0.0
All	2917	587	1458	294	9.1	1.8	171	25	6.4	1.1
Pervious surfaces										
Live	4027	752	2014	376	12.6	2.3	253	42	8.9	1.4
Dead	39	28	20	14	0.1	0.1	6	2	0.1	0.1
All	4066	763	2033	381	12.7	2.4	259	43	9.0	1.4

of yards 76-100 years old was 5716 g/m^2 more than that measured in yards 0-25 years old (Fig. 4.4). This difference, which presumably represents C that has accumulated in soils following conversion to residential use, is 2-4 times as large as average C stocks measured in trees. The difference in soil total N between new and old yards was 30-50 times as large as average N stored in trees (Fig. 4.5). Although shrubs and other vegetation were not included in this study, they likely amount to only a small fraction of total aboveground biomass. McPherson et al. (1997) found shrub C in the Chicago area to be just 4% of that found in trees, and Jo and McPherson (1995) reported an even small percentage for herbaceous plants.

Large variability in tree biomass was observed among study sites (SE = 587, Table 4.1; Fig. 4.3). The highest estimates were obtained from two unusually large parcels. Most residential parcels appear unlikely to gain such levels of aboveground biomass, since similar estimates were not observed for any of the smaller study parcels. These high estimates were retained in the dataset, however, because they represent real samples of

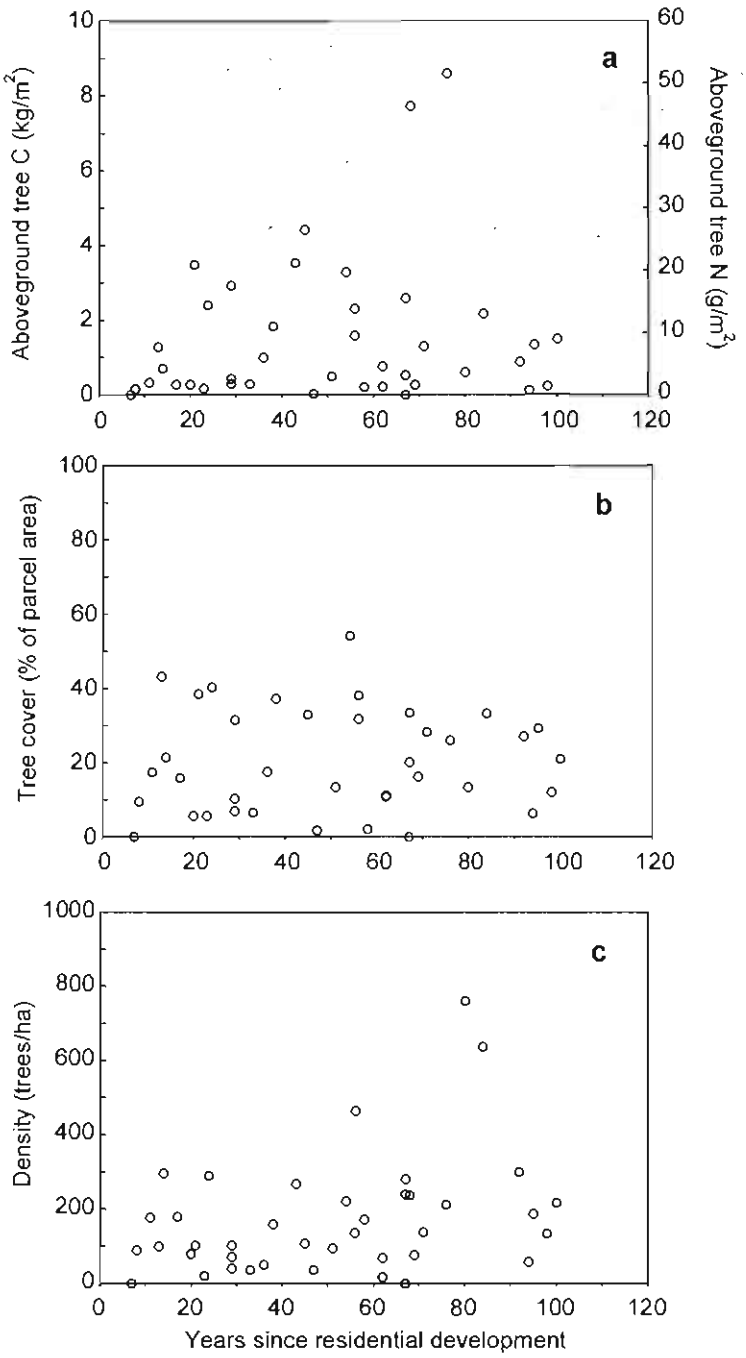

Fig. 4.3 Trends in forest structure on residential parcels as a function of yard age, including **a** aboveground tree C and N, **b** tree cover, and **c** tree density. Each marker represents a single residential parcel in the Salt Lake Valley, USA metropolitan area. All measurements shown are based on the total area of study parcels, including areas covered by both pervious and impervious surfaces. Mean tree cover was 22% and mean live tree density was 168 trees/ha.

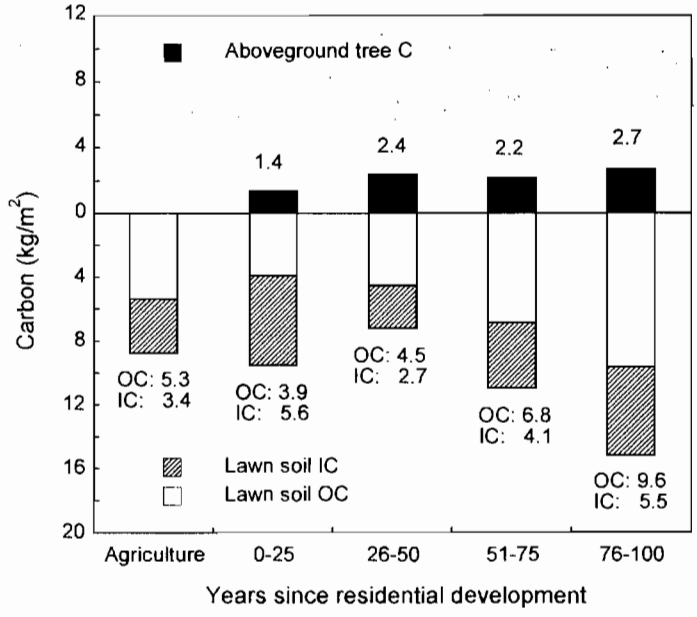

Fig. 4.4 Comparison of aboveground and belowground C storage on residential parcels in the Salt Lake Valley, USA urban ecosystem. The aboveground component includes C in biomass of live and dead trees, reported in kilogram per square meter of pervious surface. The belowground component includes soil organic carbon (SOC) and soil inorganic carbon (SIC) measured beneath lawns. Lawns comprised about 0.60 of pervious ground cover on average. Carbon stocks of individual sites were averaged by age category. $N = 3$, 10, 9, 12 and 8 for agriculture, 0-25, 26-50, 51-75 and 76-100 categories, respectively. Tree C was not measured at agricultural sites, but is assumed to be near zero.

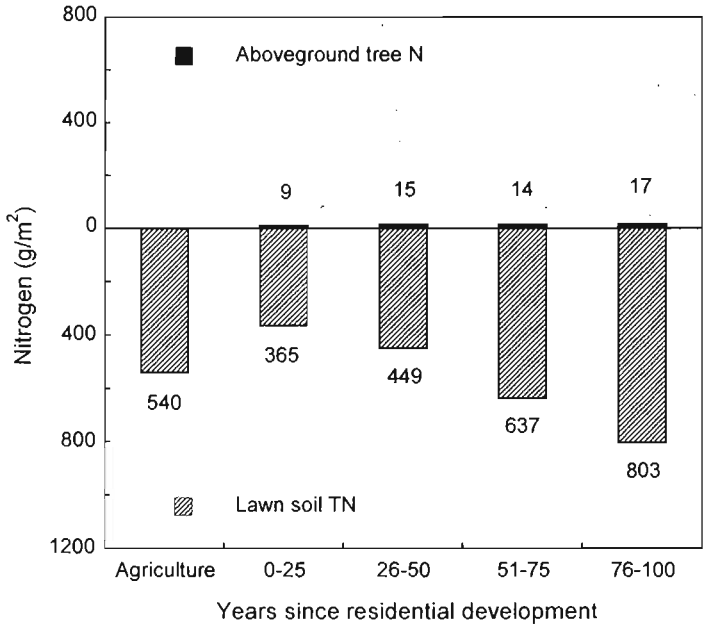

Fig. 4.5 Comparison of aboveground and belowground N storage on residential parcels in the Salt Lake Valley, USA urban ecosystem. The aboveground component includes N in biomass of live and dead trees, reported in kilograms per square meter of pervious surface. The belowground component includes soil total nitrogen (TN) to 40 cm beneath lawns. Lawns comprised about 0.60 of pervious ground cover on average. Nitrogen stocks of individual sites were averaged by age category. $N = 3$, 10, 9, 12 and 8 for agriculture, 0-25, 26-50, 51-75 and 76-100 categories, respectively. Tree N was not measured at agricultural sites, but is assumed to be near zero.

residential tree biomass in my study area. Very little of the variability in tree biomass observed among sites could be attributed to parcel age ($r^2 = 0.02$). Rather, my data suggest that at a neighborhood or regional scale, average tree biomass in the Salt Lake Valley increases quickly following residential development and reaches a nearly constant level within about three decades. The largest tree C stocks observed at study sites were comparable to or greater than SOC gains predicted in the upper 40 cm following a century of residential use. Parcels with little or no tree biomass, however, offset the large aboveground stocks observed at some sites. These findings suggest that landscaping preferences of individual homeowners exert a strong influence on average tree biomass in this study area.

Tree canopy cover and density followed similar patterns as aboveground biomass when plotted against yard age (Fig. 4.3). Two trends in particular stood out in these data: consistently high variability among sites and fairly constant average values after 10-30 years of residential use. Average tree cover on residential sites was calculated to be 22%, and average live tree density was 168 trees/ha. A reasonably strong relationship was observed between aboveground tree biomass and canopy cover on residential parcels ($r^2 = 0.70$, Fig. 4.6). This relationship, together with the availability of national and regional tree cover datasets, suggests that tree cover may be a useful parameter for mapping tree biomass across urban landscapes. This approach may be particularly feasible in arid and semiarid regions of the US, where average urban tree cover is, in general, relatively low (Nowak and Crane 2002). It is not clear from the data presented here what kind of relationship exists between canopy cover and biomass in areas where tree cover exceeds 60%. Indeed, biomass will likely be grossly overestimated if the exponential function

developed from these data is applied to areas of dense tree cover. Although tree cover measured in this study is highly representative of that found in the Salt Lake Valley urban ecosystem as a whole (Fig. 4.7), more work is needed to characterize denser urban forests and to validate the accuracy of using canopy cover to estimate urban tree biomass.

Residential forest structure

Aboveground biomass on residential parcels is influenced by both tree number and tree size. On an individual tree basis, biomass increases exponentially with increasing diameter (Nowak 1994). At my sites, trees larger than 40 cm DBH accounted for just 10% of the total tree count but 66% of the total tree biomass (Fig. 4.8). Trees larger than 70 cm DBH alone represented 33% of total biomass. It is possible that allometric equations available in the literature overestimate the biomass of large trees, since the majority of these equations were not developed using trees larger than 70 cm. Nevertheless, residential urban forests of the Salt Lake Valley appear to consist primarily of small diameter trees, while large diameter trees appear to contribute most to total biomass. Sixty-seven percent of all trees encountered on study sites were smaller than 20 cm DBH. The total number of trees in each size class decreased progressively with increasing diameter. The latter trend was also observed in other US cities (Nowak 1994; Jo and McPherson 2001) and may be even more pronounced in native forests. Trees smaller than 12.7 cm in diameter comprise 66% of all live trees in native Utah forests (O'Brien 1999).

Small diameter trees greatly outnumbered large trees across the entire residential age gradient (Fig. 4.9). This trend, together with the lack of significant changes in average biomass beyond the first 30 years, suggests that tree mortality and/or active

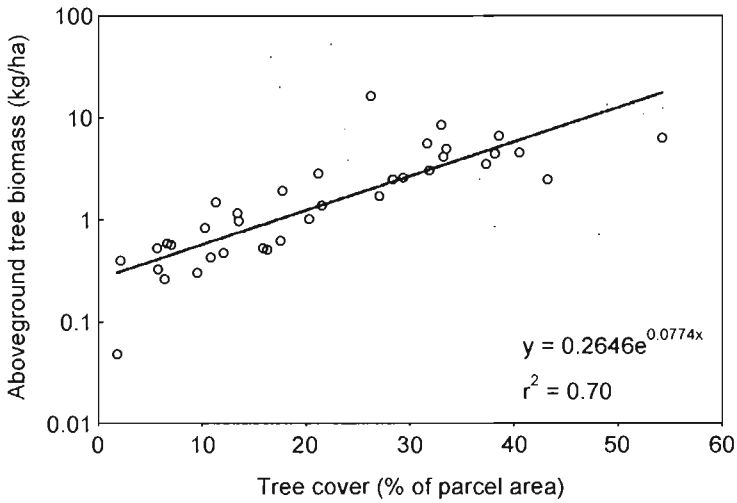

Fig. 4.6 Relationship between tree canopy cover and aboveground biomass on residential parcels. Each marker represents a single study site. Canopy cover and biomass measurements shown are based on the total area of study parcels, including areas covered by both pervious and impervious surfaces.

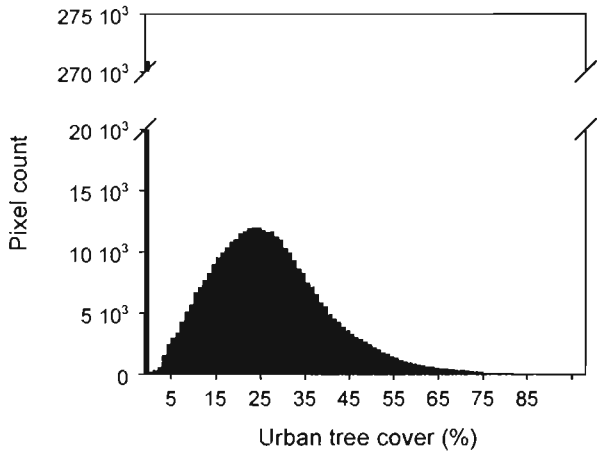

Fig. 4.7 Histogram of urban tree cover in the Salt Lake Valley derived from the 2001 National Land Cover Dataset (Homer et al. 2007). Urban tree cover was estimated by clipping the original 2001 NLCD tree cover dataset to only those pixels classified as urban land cover. Mean tree cover was 16% for the entire urban area and 28% for those areas with at least 1% cover.

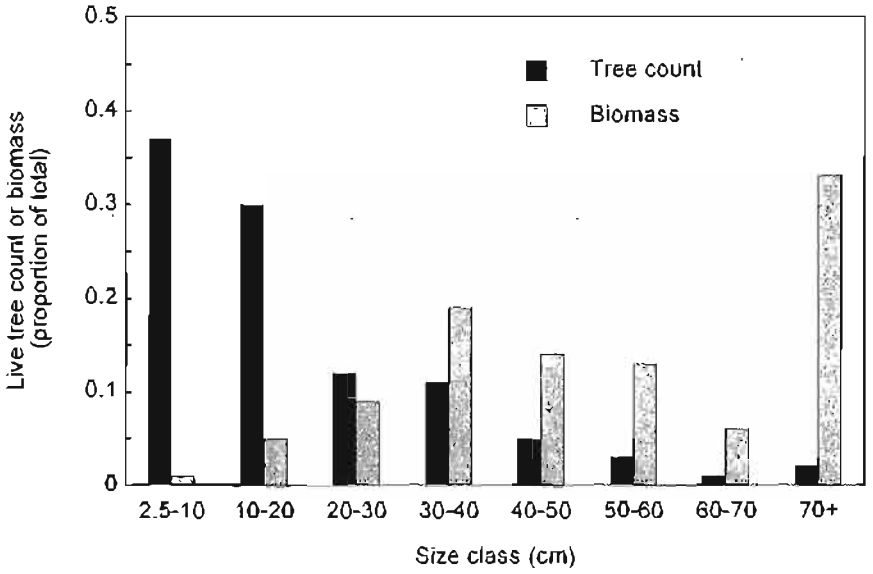

Fig. 4.8 Distribution of tree count and biomass by diameter size class.

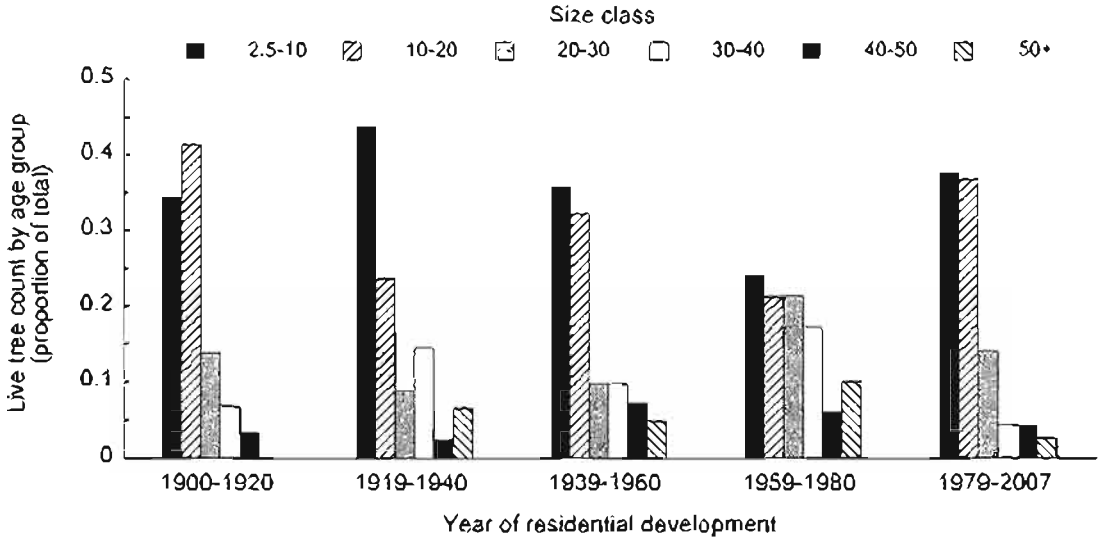

Fig. 4.9 Tree count by diameter class and age of residential parcel.

human management is strongly limiting the persistence of large diameter trees. My data imply that approximately 28% of trees less than 10 cm DBH will reach diameters greater than 40 cm. Tree survival in unmanaged forests may be even lower than that reported here, since urban areas typically have a greater proportion of large trees than do native forests (Nowak and Crane 2002). While tree mortality can be substantial in natural settings (O'Brien 1999), human influence no doubt plays a heightened role in the urban environment. Most trees in the residential neighborhoods of the Salt Lake Valley appear to be removed prior to or promptly following tree death, since dead trees accounted for just 2.3% of total tree count and 1.1% of total biomass.

Hardwood species dominated the residential urban forests sampled in this study (Fig. 4.10). About half of the 65 species identified are not native to North America, and less than 20% are native to Utah (Fig. 4.11). Nevertheless, Utah natives accounted for more than one-third of all live trees measured, due in large part to the abundance of quaking aspen (*Populus tremuloides*), a popular species that re-sprouts extensively from its roots. Quaking aspen occurred at more than one-third of all study sites and alone represented 21% of the total tree count (Table 4.2). This species contributed minimally to total tree biomass of residential parcels, however, due to its consistently small size. Siberian elm (*Ulmus pumila*) accounted for more total tree biomass than any other species and was the third most common species encountered at study sites. This Asian native is highly naturalized in urban portions of the Salt Lake Valley, where it becomes established in unmanaged areas and can grow to considerable size. Other common species contributing notably to total study biomass were Lombardy poplar (*Populus*

Fig. 4.10 Proportion of live trees that were gymnosperm or angiosperm.

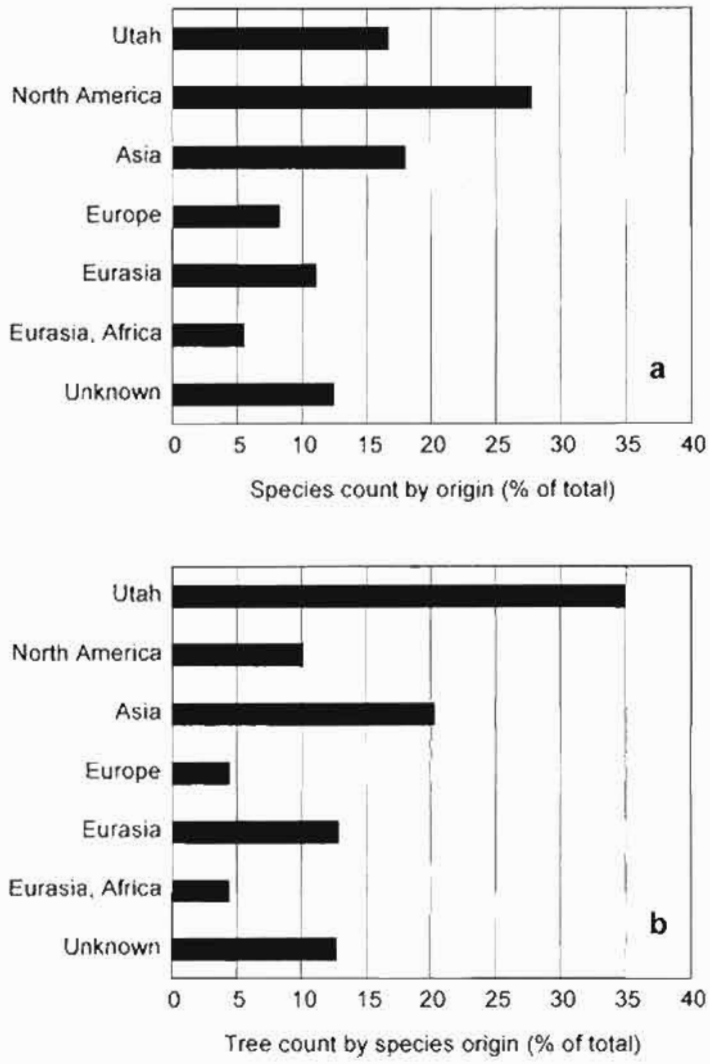

Fig. 4.11 Origin of tree species measured on residential parcels. The North America category does not include species native to Utah.

nigra), blue spruce (*Picea pungens*), Norway maple (*Acer platanoides*), and box elder (*Acer negundo*).

Of all trees measured at my sites, 61% were angiosperm species with diffuse porous wood anatomy (Fig. 4.10). Ring porous species represented 19% of the total tree count, and the remaining trees were gymnosperms or unidentified species. As a consequence of this unbalanced distribution, diffuse porous species contributed approximately three times more biomass and basal area, on average, than ring porous species (Table 4.3).

Comparisons with other cities

Intensive inventories of urban forests have been conducted for at least 15 other US cities (Nowak and Crane 2002; Nowak et al. 2005; Golubiewski 2006). Most of these studies were designed to provide unbiased estimates of tree biomass and other forest characteristics for entire metropolitan areas, rather than to investigate changes in urban forest structure over long time periods (although see Golubiewski 2006). In most cases, study plots were fixed in size and randomly located within different urban land use strata (Nowak and Crane 2000). As a result, information from other US urban forests may not be directly comparable to that presented here. My estimates of tree biomass, density and canopy cover, for example, do not account for areas occupied by streets, parking lots and other features outside the boundaries of residential parcels. Nor do they account for contributions of street trees to biomass and canopy cover. Furthermore, lands committed to commercial, institutional and recreational uses within the Salt Lake Valley metropolitan area almost certainly differ from residential neighborhoods with regard to average tree biomass, density and canopy cover.

It is nonetheless interesting to place my findings within the context of those from

Table 4.2 Ten most dominant tree species measured on residential parcels (live trees only)

By number	% Total count	% Total biomass	By biomass	% Total count	% Total biomass
quaking aspen *	20.8	2.1	Siberian elm	6.4	15.9
cherry/chokecherry/plum [u]	10.2	2.6	weeping willow	0.3	12.9
Siberian elm	6.4	15.9	Lombardy poplar	3.3	11.3
Norway maple	5.4	4.1	white poplar	3.8	8.4
blue spruce *	5.1	8.4	blue spruce *	5.1	8.4
box elder *	4.1	3.0	Norway maple	5.4	4.1
white poplar	3.8	8.4	unknown willow spp. [u]	0.2	3.7
Austrian pine	3.5	2.6	globe willow	1.1	3.4
Lombardy poplar	3.3	11.3	box elder *	4.1	3.0
Oriental arborvitae	2.9	0.5	Russian olive	2.7	3.0
Totals	65.5	58.9	Totals	32.4	74.1
All other species	34.5	41.1	All other species	67.6	25.9

* Utah native, [u] May include species both native and not native to Utah

Table 4.3 Average live aboveground tree biomass, carbon storage, nitrogen storage, density and basal area of residential parcels, reported by vessel type

	Biomass (g/m^2)		Carbon (g/m^2)		Nitrogen (g/m^2)		Density (trees/ha)		Basal area (m/ha)	
	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE
Entire parcel										
Diffuse porous	1813	439	906	219	5.7	1.4	115	22	3.8	0.7
Ring porous	610	236	305	118	1.9	0.7	23	6	1.3	0.5
Pervious surfaces										
Diffuse porous	2615	614	1307	307	8.2	1.9	178	37	5.6	1.0
Ring porous	789	271	394	135	2.5	0.8	33	8	1.7	0.5

other cities. Nowak and Crane (2002) estimated C storage of urban forests in each state using field data from 10 US cities and national urban tree cover estimates. They estimated C storage of Utah's urban forests to be 13.0 t/ha, which included C stored in both aboveground and belowground biomass. In comparison, aboveground tree C stocks on my residential sites averaged 14.6 t/ha (Table 4.1). Applying the root-to-shoot ratio used by Nowak and Crane (2002) of 0.26, my estimate becomes 18.4 t/ha for total aboveground and belowground tree C. This value is still considerably lower than the national average of 25.1 t/ha estimated for all urban land cover types, but higher than the same estimate made for just the Rocky Mountain region (16.1 t/ha, Nowak and Crane 2002). Average tree C storage in residential neighborhoods of the Chicago area was determined to be 22.4 t/ha (Nowak 1994).

C sequestration by residential trees

With regard to the 40 residential sites sampled in this study, the relatively rapid rise and plateau of average tree biomass and canopy cover is striking. Unlike gains in lawn SOC, which may occur in the study region for longer than a century (Chapter 2), net C sequestration of urban forests appears to be limited on average to the first few decades following urbanization. After this initial period, C lost through tree respiration and decomposition presumably balances C gained through photosynthesis. C emissions related to tree maintenance and irrigation may subsequently convert these forests from net sinks of atmospheric C to net sources (Nowak et al. 2002).

Tree cover is a conspicuous and novel component of the modern Salt Lake Valley ecosystem. Although trees may provide a variety of services to urban residents, direct C sequestration by urban trees likely offsets only a small fraction of fossil C emitted by a

city. For an average residential parcel sampled in this study, existing aboveground tree C stocks amounted to less than 30% of annual CO_2e emissions currently produced by the average Utah resident (Roe 2007). CO_2e includes all greenhouse gas emissions expressed in terms of CO_2 based on the radiative forcing of each gas relative to CO_2. Nowak (1994) reported that total tree C stocks in the Chicago metropolitan area equated to just 5 months of C emissions by the residential sector, including emissions associated with transportation. Aboveground C sequestration by residential trees in my study area appears to be about 0.49 t ha^{-1} yr^{-1}, assuming that steady state conditions are reached after 30 years and that no trees were present prior to urban development (average aboveground tree C divided by 30). This value is very close to the gross sequestration rate of 0.4 t ha^{-1} yr^{-1} estimated for Utah urban forests by Nowak and Crane (2002).

Average C stored per hectare on Salt lake Valley residential parcels is greater than that reported for native Utah forests, when only aboveground live tree biomass is considered (USDA Forest Service 2008b). Residential stocks are nearly twice those reported for the state when based on pervious surfaces only. Urban areas in Utah comprise 3.5% of the area covered in forests and thus have contributed minimally to changes in Utah forest extent over the past century (Nowak and Crane 2002; USDA Forest Service 2008b). Expansion of pinyon and juniper woodlands over the same period has likely added considerably more forest acreage. Woodlands dominated by these short-statured tree species now comprise nearly 50% of the total forested area in Utah and contribute to the relatively low average biomass of forests in the state (O'Brien 1999).

Direct storage of atmospheric C is, of course, only one service provided by urban forests. Urban trees may also indirectly reduce atmospheric CO_2 levels by influencing

building energy efficiency. Trees planted on the west and east sides of structures can lessen summer air-conditioner use through shading and evaporative cooling. Strategically placed trees can also reduce wind speeds around urban dwellings, resulting in lower winter heating costs. In two residential blocks in Chicago, Jo and McPherson (2001) estimated that trees reduced C emissions by –0.2 to 3.9% through shading, evapotranspiration and wind reduction. A 10% increase in tree cover in Sacramento, CA and Phoenix, AZ could potentially reduce cooling energy by 24 and 12%, respectively (Huang et al. 1992). Although studies suggest that indirect C reductions by trees may greatly exceed those resulting from direct C storage, these reductions are still very modest relative to anthropogenic C emissions. It has been estimated that planting 100 million trees in US cities in energy-saving locations would result in rates of C avoidance four times larger than direct C sequestration by trees (Nowak and Crane 2002). In the 50 years following planting, both forms of C reductions would equate to less than 1% of estimated C emissions in the US (Nowak and Crane 2002).

Conclusions

Urbanization of the Salt Lake Valley has resulted in an increase in tree cover, from near zero prior to European settlement to about 16% tree cover in 2001. This study attempted to quantify changes in ecosystem C and N stocks associated with this urban afforestation. Among residential study parcels varying in age from 7 to 100 years, I observed no significant relationship between aboveground tree biomass and time since residential development ($P < 0.05$). This finding is distinct from trends reported in earlier chapters between yard age and soil organic C and total N content. Aboveground tree biomass was highly variable among study sites. Nevertheless, my data suggest that

average aboveground tree biomass on Salt Lake Valley residential parcels reaches nearly steady levels within about three decades following urban development. Aboveground tree biomass was a function both of tree number and average tree size. Residential urban forests sampled in this study were dominated by small diameter trees, while large diameter trees contributed most to total biomass. Such trends held true regardless of neighborhood age.

Average aboveground tree C on residential sites was estimated to be 1458 g/m^2 for entire parcels and 2033 g/m^2 when calculated using pervious surfaces only. These stocks are smaller than average SOC gains predicted for the upper 40 cm of residential lawns following a century of residential use. They are, however, of the same order of magnitude. For an average residential parcel sampled in this study, these tree C stocks amounted to less than 30% of annual CO_2e emissions currently produced by an average Utah resident (Roe et al. 2007). Direct carbon sequestration by urban trees likely offsets only a small fraction of anthropogenic C emissions in the region, but this process may gain significance when combined with belowground increases in C stocks and indirect C reductions through shading, wind reduction and evaporative cooling.

References

Chojnacky DC, Rogers P (1999) Converting tree diameter measured at root collar to diameter at breast height. Western Journal of Applied Forestry 14:14-16

Golubiewski NE (2006) Urbanization transforms prairie carbon pools: effects of landscaping in Colorado's Front Range. Ecological Applications 16:555-571

Homer C, Dewitz J, Fry J, Coan M, Hossain N, Larson C, Herold N, McKerrow A, VanDriel JN, Wickham J (2007) Completion of the 2001 National Land Cover Database for the conterminous United States. Photogrammetric Engineering and Remote Sensing 73: 337-341

Huang J, Ritschard R, Sampson N, Taha H (1992) The benefits of urban trees. In: Akbari H, Davis S, Dorsano S, Huang J, Winnett S (eds) Cooling our communities. US Environmental Protection Agency, Washington, DC, pp 27-42

Jenkins JC, Chojnacky DC, Heath LS, Birdsey RA (2003) National -scale biomass estimators for United States tree species. Forest Science 49:12-35

Jenkins JC, Chojnacky DC, Heath LS, Birdsey RA (2004) Comprehensive database of diameter-based biomass regressions for North American tree species. USDA Forest Service Gen Tech Rep NE-319

Jo H, McPherson EG (1995) Carbon storage and flux in urban residential greenspace. Journal of Environmental Management 45:109-133

Jo H-K, McPherson EG (2001) Indirect carbon reduction by residential vegetation and planting strategies in Chicago, USA. Journal of Environmental Management (2001) 61:165-177

McPherson EG, Nowak D, Heisler G, Grimmond S, Souch C, Grant R, Rowntree R (1997) Quantifying urban forest structure, function, and value: thc Chicago Urban Forest Climate Project. Urban Ecosystems 1:49-61

Nowak DJ (1994) Atmospheric carbon dioxide reduction by Chicago's urban forest. In: McPherson EG, Nowak DJ, Rowntree, RA (eds) Chicago's urban forest ecosystem: results of the Chicago Urban Forest Climate Project. USDA Forest Service General Technical Report NE-186

Nowak DJ, Crane DE (2002) Carbon storage and sequestration by urban trees in the USA. Environmental Pollution 116:381-389

Nowak DJ. Hoehn RE III, Harte A, Stevens JC, Walton JT, Crane DE, Brandy L (2005) Assessing urban forest effects and values. USDA Forest Service Northeastern Research Station General Technical Report

Nowak DJ, Stevens JC, Sisinni SM et al. (2002) Effects of urban tree management and species selection on atmospheric carbon dioxide. Journal of Arboriculture 28:113-122

O'Brien RA (1999) Comprehensive inventory of Utah's forest resources, 1993. USDA Forest Service Resource Bulletin RMRS-RB-1

Roe S, Strait R, Bailie A, Lindquist H, Jamison A (2007) Final Utah greenhouse gas inventory and reference case projections, 1900-2020. Center for Climate Strategies, Utah Department of Environmental Quality

Schlesinger WH (1991) Biogeochemistry: an analysis of global change, edition. Academic Press, Inc. San Diego, CA

Ter-Mikaelian MT, Korzukhin MD (1997) Biomass equations for sixty-five North American tree species. Forest Ecology and Management 97:1-24

USDA Forest Service (2008a) Publication and inventory results. Interior West Forest Inventory and Analysis Program, http://www.fs.fed.us/rm/ogden/publications/index

USDA Forest Service (2008b) Annual tables. Interior West Forest Inventory and Analysis Program, http://www.fs.fed.us/rm/ogden/annual_tables

Wakefield JH (1933) A study of the plant ecology of the Salt Lake and Utah Valleys before the Mormon immigration. MS thesis, Brigham Young University, 56 pp

Zianis D, Muukkonen P, Mäkipää R, Mencuccini M (2005) Biomass and stem volume equations for tree species in Europe. Silva Fennica Monographs 4:1-63

APPENDIX A

SOIL MAPS OF THE SALT LAKE VALLEY, USA

Fig. A.1 Spatial variation in soil organic carbon (SOC) within the Salt Lake Valley, derived from a regional NRCS SSURGO database. SOC was mapped for depths of both **a** 0-20 cm and **b** 0-40 cm.

Soil organic carbon was calculated from SSURGO using a method similar to Bliss et al. (1995). The SSURGO database maps soils according to discrete and spatially exhaustive "map units". Each map unit consists of one or more "component". For each component, generalized data are reported by horizon. I initially calculated SOC for each individual horizon of each component. For those horizons (or portions of horizons) falling between 0-20 or 0-40 cm, horizon values were then summed to derive component SOC at the two depths. Map unit SOC was calculated using a weighted average that takes into account the relative contribution of a component to a particular map unit.

The NRCS reports high(h), low(l) and representative(r) values of organic matter, bulk density, and percentage rock and soil fractions. Only representative(r) values were used in calculations here.

$$OC_{horizon}\ (g\ m^{-2}) = \%OM/100 * 0.58 * \text{bulk density}_{oven\ dry} * R * no10 * depth * 10\ 000 \quad (A.1)$$

$$OC_{component}\ (g\ m^{-2}) = \sum OC_{horizon} \quad (A.2)$$

$$OC_{map\ unit}\ (g\ m^{-2}) = (\sum OC_{component} * \%contribution) / \sum \%contribution \quad (A.3)$$

where R = Rock fragment conversion factor, calculated as follows:

$$R = (F/\text{bulk density}_{oven\ dry})/((F/\text{bulk density}_{oven\ dry})+(1-F)/2.65)) \quad (A.4)$$

$$\text{where } F = (1- inch10/100- inch3/100)*no10/100 \quad (A.5)$$

inch10 = rock fragments >250 mm
inch3 = rock fragments 75–250 mm
no10 = >2.0 mm fraction from that which has passed through a 75-mm sieve

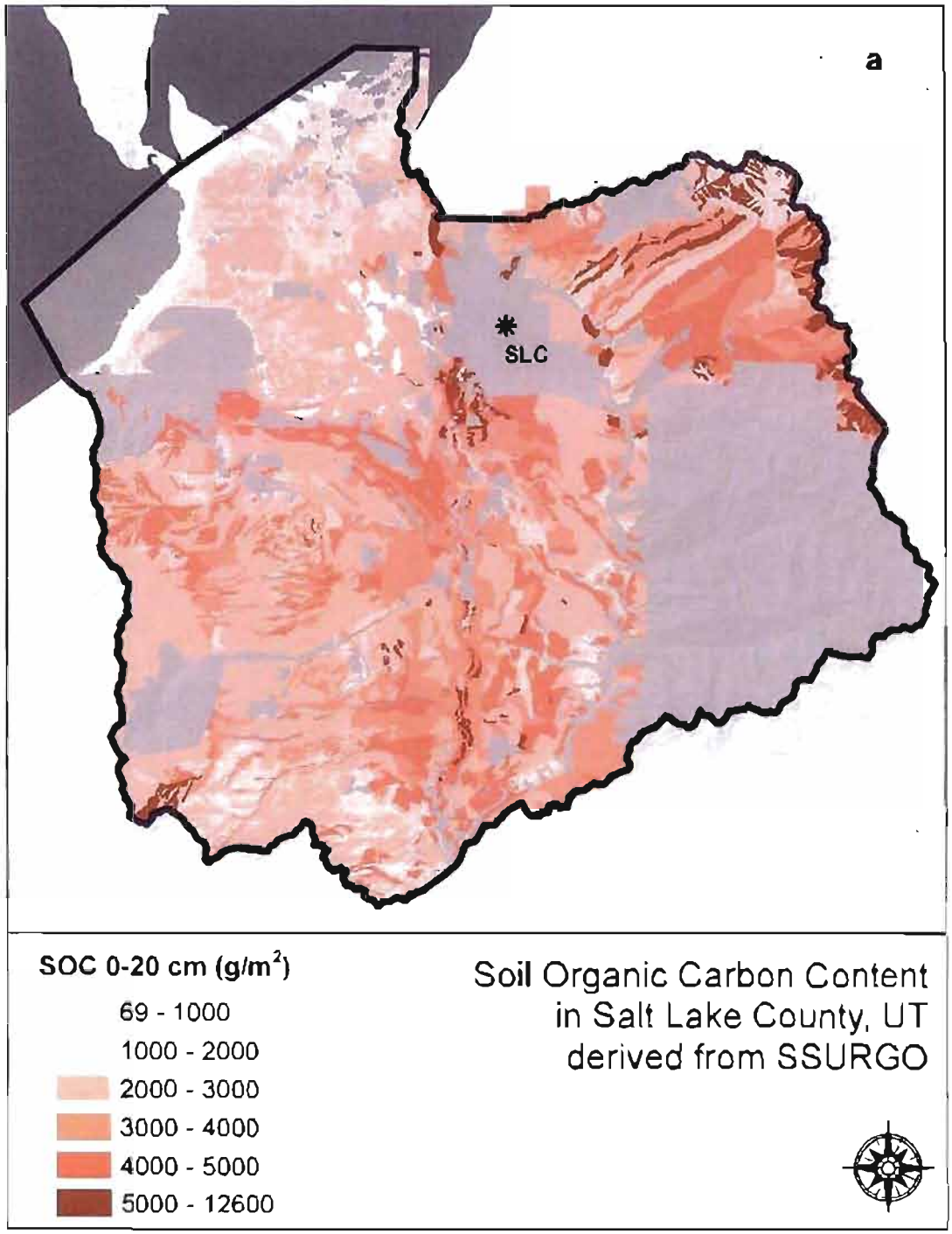

Fig. A.1 continued

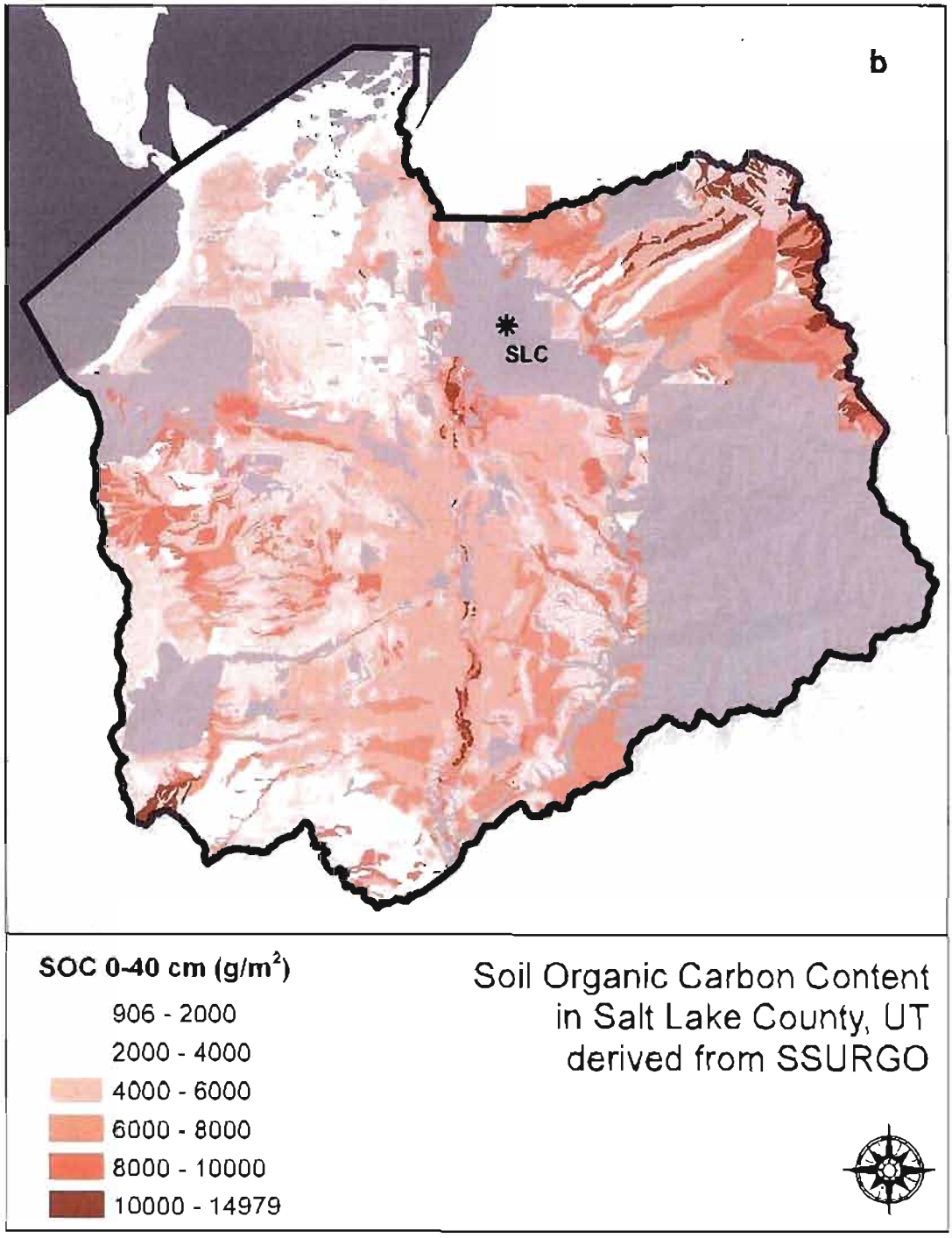

Fig. A.1 continued

Figure A.2 Spatial variation in soil texture within the Salt Lake Valley, derived from a regional NRCS SSURGO database. Maps of **a** % sand content and **b** % clay content are shown.

Percent clay and sand were calculated for each map component as a weighted average of horizon values in the upper 40 cm of the soil profile. I calculated map unit soil texture using a weighted average that takes into account the relative contribution of a component to a particular map unit.

$$\%Clay_{component} = (\sum \%Clay_{horizon} * depth_{horizon}) / \sum depth_{horizon} \quad (A.6)$$

$$\%Clay_{map\ unit} = (\sum \%Clay_{component} * \%contribution) / \sum \%contribution \quad (A.7)$$

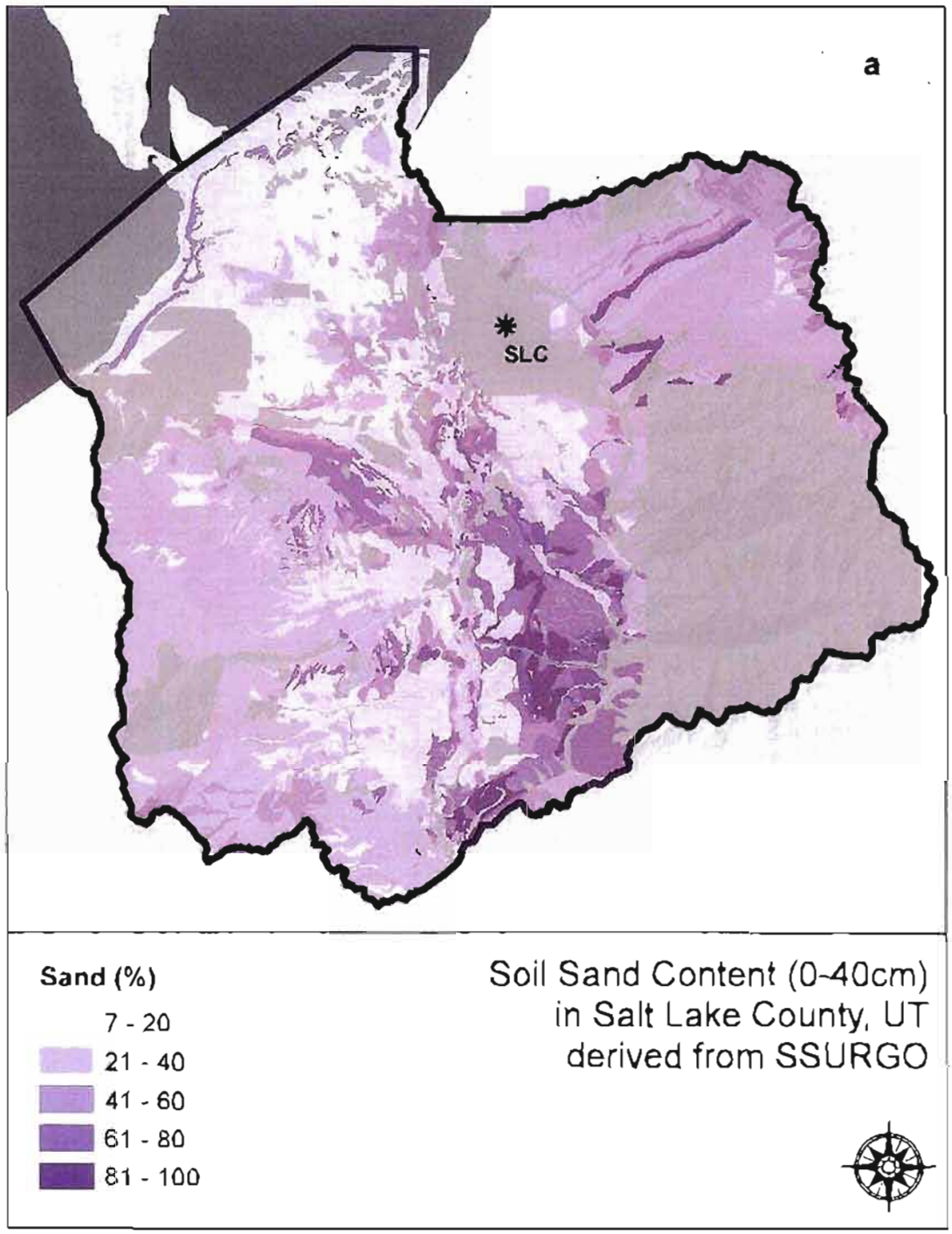

Fig. A.2 continued

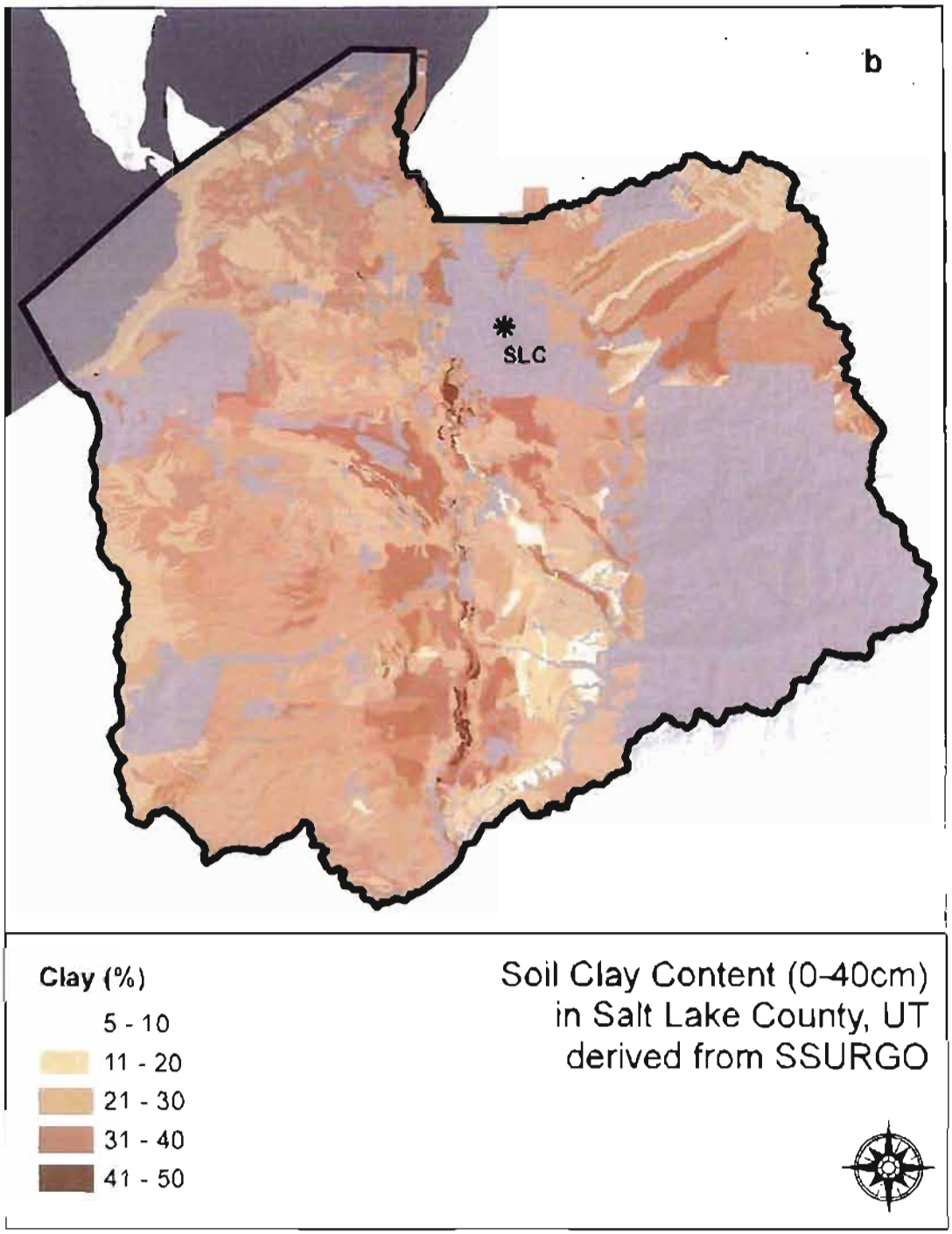

Fig. A.2 continued

References

Bliss NB, Waltman SW, Peterson GW (1995) Preparing a soil carbon inventory for the United States using Geographic Information Systems. In Lal R, Kimble J, Levine E, Stewart BA (eds) Soils and global change. Lewis Publishers, Boca Raton, London, Tokyo

APPENDIX B

SUPPLEMENTAL TREE TABLES AND FIGURES

Table B.1 Mean tree measurements of hardwood species by size class (live trees only)

Size class	Count	DBH (cm)	Height (m)	Crown height (m)	Crown width (m)	Aboveground biomass (kg)
2.5-10	196	5.9	5.2	4.1	2.0	7.1
10-20	153	13.9	8.1	6.4	3.7	47.9
20-30	63	24.9	9.5	7.7	5.4	194.2
30-40	57	34.9	12.8	9.6	7.1	435.8
40-50	18	45	18.1	14.9	7.8	808.8
50-60	10	54.4	17.8	14.8	10.4	1237.2
60-70	5	63.0	21.9	19.1	9.8	1867.8
70+	10	100.0	20.5	17.8	12.7	5050.3
All	512	18.6	8.6	6.6	4.0	258.9

Table B.2 Mean tree measurements of softwood species by size class (live trees only)

Size class	Count	DBH (cm)	Height (m)	Crown height (m)	Crown width (m)	Aboveground biomass (kg)
2.5-10	35	5.8	3.3	3.2	1.6	6.8
10-20	33	13.2	4.1	4.0	2.1	32.0
20-30	15	25.1	6.7	6.2	3.8	148.4
30-40	13	35.1	11.2	9.7	6.7	353.3
40-50	12	43.6	15.1	12.7	6.7	595.6
50-60	8	54.3	13.5	12.5	7.9	1060.5
60-70	0					
70+	1	75.5	15.1	13.8	10.4	2403.6
All	117	21.4	6.9	6.2	3.6	223.5

Table B.3 Mean tree measurements of all species by size class (live trees only)

Size class	Count	DBH (cm)	Height (m)	Crown height (m)	Crown width (m)	Aboveground biomass (kg)
2.5-10	231	5.9	4.9	3.9	2.0	7.1
10-20	186	13.8	7.4	5.9	3.4	45.1
20-30	78	24.9	9.0	7.4	5.1	185.4
30-40	70	34.9	12.5	9.6	7.0	420.5
40-50	30	44.4	16.9	14.0	7.4	723.5
50-60	18	54.3	15.9	13.6	9.1	1158.7
60-70	5	63.0	21.9	19.1	9.8	1867.8
70+	11	97.8	20.0	17.3	12.4	4809.6
All	629	19.1	8.3	6.5	3.9	252.3

Table B.4 Mean live tree measurements of diffuse porous species by size class

Size class	Count	DBH (cm)	Height (m)	Crown height (m)	Crown width (m)	Aboveground biomass (kg)
2.5-10	158	5.8	5.2	3.9	2.0	7.1
10-20	123	13.9	8.1	5.9	3.6	47.4
20-30	47	24.5	9.9	7.4	5.5	188.3
30-40	28	34.9	13.3	9.6	7.4	467.1
40-50	12	45.6	19.0	14.0	8.0	827.4
50-60	6	54.7	18.1	13.6	9.7	1275.4
60-70	4	63.5	22.8	19.1	10.5	1770.8
70+	7	99.1	18.6	17.3	12.7	5096.5
All	385	17.1	8.3	6.5	3.8	231.7

Table B.5 Mean live tree measurements of ring porous species by size class

Size class	Count	DBH (cm)	Height (m)	Crown height (m)	Crown width (m)	Aboveground biomass (kg)
2.5-10	37	6.0	5.4	3.9	2.3	7.4
10-20	23	13.9	8.2	5.9	4.4	50.3
20-30	16	26.1	8.4	6.9	5.1	211.8
30-40	29	34.9	12.3	8.8	6.6	405.6
40-50	6	43.7	16.4	11.7	7.6	771.5
50-60	4	53.9	17.4	13.3	11.1	1179.9
60-70	1	60.8	18.3	10.3	7.6	2256.0
70+	3	102.3	25.0			4942.4
All	119	23.7	9.6	6.5	4.6	361.5

Table B.6 Tree species measured on residential study sites (live trees only)

Species	Common name	Count	% Total	No. of sites	Native range
Abies concolor	white fir	1	0.2	1	Utah
Acer grandidentatum	bigtooth maple	2	0.3	1	Utah
Acer negundo	box elder	26	4.1	6	Utah
Acer palmatum	Japanese maple	1	0.2	1	Asia
Acer platanoides	Norway maple	34	5.4	12	Eurasia
Acer pseudoplatanus	planetree maple	1	0.2	1	N. America
Acer rubrum	red maple	4	0.6	3	N. America
Acer saccharinum	silver maple	1	0.2	1	N. America
Aesculus hippocastanum	horse chestnut	1	0.2	1	Europe
Ailanthus altissima	tree-of-heaven	13	2.1	3	Asia
Betula occidentalis	water birch	1	0.2	1	Utah
Betula papyrifera	paper birch	9	1.4	4	N. America
Betula pendula	European white birch	1	0.2	1	Eurasia
Catalpa speciosa	northern catalpa	1	0.2	1	N. America
Celtis occidentalis	common hackberry	2	0.3	1	N. America
Cercis canadensis	eastern rosebud	1	0.2	1	N. America
Cercocarpus ledifolius	curl-leaf mtn mahogany	5	0.8	1	N. America
Crataegus laevigata	English hawthorn	1	0.2	1	Eurasia, Africa
Elaeagnus angustifolia	Russian olive	17	2.7	1	Asia
Fagus sylvatica	European beech	1	0.2	1	Europe
Fraxinus americana	white ash	4	0.6	2	N. America
Fraxinus pennsylvanica	green ash	5	0.8	4	Utah
Ginkgo biloba	ginkgo	3	0.5	2	Asia
Gleditsia triacanthos	honey locust	14	2.2	8	N. America
Gymnocladus dioicus	Kentucky coffee tree	1	0.2	1	N. America
Juglans nigra	black walnut	2	0.3	2	N. America
Juniperus chinensis	Chinese juniper	10	1.6	1	Asia
Juniperus scopulorum	Rocky Mountain juniper	3	0.5	1	Utah
Juniperus sp.	juniper	1	0.2	1	Unknown
Laburnum watereri	golden chain tree	2	0.3	1	Europe
Liriodendron tulipifera	tulip tree	1	0.2	1	N. America
Magnolia soulangiana	saucer magnolia	1	0.2	1	Unknown
Magnolia sp.	magnolia	1	0.2	1	Unknown
Malus pumila	common apple	6	1.0	5	Eurasia
Malus sp.	crabapple	3	0.5	3	Unknown
Morus alba	white mulberry	2	0.3	1	Asia
Morus rubra	red mulberry	1	0.2	1	N. America

Table B.6 continued

Species	Common name	Count	% Total	No. of sites	Native range
Picea abies	Norway spruce	1	0.2	1	Europe
Picea glauca	white spruce	2	0.3	1	N. America
Picea glauca (dwarf)	white spruce (dwarf)	10	1.6	5	N. America
Picea pungens	blue spruce	32	5.1	11	Utah
Picea sp.	spruce	2	0.3	2	Unknown
Pinus contorta	lodgepole pine	1	0.2	1	Utah
Pinus nigra	Austrian pine	22	3.5	9	Europe
Pinus ponderosa	ponderosa pine	5	0.8	2	Utah
Pinus sylvestris	Scots pine	5	0.8	3	Eurasia
Platanus occidentalis	American sycamore	2	0.3	1	N. America
Platycladus orientalis	Oriental arborvitae	18	2.9	6	Asia
Populus alba	white poplar	24	3.8	1	Eurasia
Populus nigra	Lombardy poplar	21	3.3	5	Eurasia, Africa
Populus sp.	poplar	2	0.3	1	Unknown
Populus tremuloides	quaking aspen	131	20.8	14	Utah
Prunus armeniaca	apricot	1	0.2	1	Asia
Prunus persica	nectarine/peach	6	1.0	3	Asia
Prunus sp.	cherry/chokecherry/plum	64	10.2	20	Unknown
Pseudotsuga menziesii	Douglas fir	2	0.3	2	Utah
Pyrus calleryana	Bradford pear	9	1.4	6	Asia
Pyrus communis	common pear	6	1.0	4	Eurasia
Quercus gambelii	Gambel oak	11	1.7	3	Utah
Quercus macrocarpa	burr oak	1	0.2	1	N. America
Quercus palustris	pin oak	1	0.2	1	N. America
Salix matsudana	globe willow	7	1.1	6	Asia
Salix sepulcralis	weeping willow	2	0.3	1	Eurasia
Salix sp.	willow	1	0.2	1	Unknown
Sorbus aucuparia	European mountain ash	3	0.5	2	Eurasia
Tamarisk sp.	tamarisk	5	0.8	1	Eurasia, Africa
Taxus baccata	English yew	1	0.2	1	Eurasia, Africa
Thuja plicata	western redcedar	1	0.2	1	N. America
Tilia cordata	common linden	1	0.2	1	Europe
Ulmus pumila	Siberian elm	40	6.4	6	Asia
unknown hardwood	unknown hardwood	5	0.8	2	Unknown
Zelkova serrata	Japanese zelkova	1	0.2	1	Asia

Table B.7 Average tree measurements by species (live trees only)

Species	DBH (cm)	Height (m)	Crown height (m)	Crown width (m)	Biomass (kg)	Biomass equation reference(s)
Abies concolor	16.2	5.5	4.9	2.6	63.4	28
Acer grandidentatum	12.5	11.8	9.8	3.3	38	11
Acer negundo	18.7	8.5	7.1	3.8	185.1	26
Acer palmatum	4.2	4.4	3.5	2.9	3.5	26
Acer platanoides	18.2	9.2	7.3	5.2	189.1	26
Acer pseudoplatanus	32.8	14.1	11.1	8.5	454.8	26
Acer rubrum	11.4	5.9	4.5	4	52.8	5,6,16,18,22,34,35,36, 37,42,45,46,49,51
Acer saccharinum	22.6	11.2	10.1	7.3	188.4	26
Aesculus hippocastanum	49.2	13.5	13.4	8.6	1066.7	25
Ailanthus altissima	12.9	8.7	5.4	3.7	81.9	25
Betula occidentalis	13.4	5.3	3.8	3.3	54.7	26
Betula papyrifera	8.3	7.3	6.4	3.2	33.4	3,18,34,35,37,42,46,4 7,48,51
Betula pendula	14.9	9.5	8.9	4.4	64.8	30
Catalpa speciosa	6.8	4.7	3.3	3	7.8	25
Celtis occidentalis	8.5	5.4	4.5	2.6	15	25
Cercis canadensis	3.1	3.4	2.9	3.2	1.1	25
Cercocarpus ledifolius	7.4	4.3	3.9	3.1	2.8	10
Crataegus laevigata	31.7	8.6	6.9	7.8	358	25
Elaeagnus angustifolia	28.7	5.7	4.7	3.9	282.3	25
Fagus sylvatica	3.5	3.2	2.6	1.5	3.5	2,13
Fraxinus americana	31.2	11.7	6.8	4.9	877.8	5,22,33,42
Fraxinus pennsylvanica	26.2	10.4	8	5.9	346.5	14
Ginkgo biloba	8.4	6.1	5.9	3.1	17.4	25
Gleditsia triacanthos	14.8	7.4	5.9	5.3	153.9	25
Gymnocladus dioicus	52.6	19.4	15.7	12	1259.2	25
Juglans nigra	20.8	8.5	7.5	7.7	251	25
Juniperus chinensis	8.7	3.9	3.8	1.8	11.6	19
Juniperus scopulorum	4.5	4.4	4.2	1.1	5.6	19
Juniperus sp.	4.7	5.6	4.9	2.6	5.3	19
Laburnum watereri	8.5	5.8	4.7	1.8	13.9	5,25,40,49,50
Liriodendron tulipifera	6.4	4.1	3.5	2.2	3.8	5, 40, 43, 49, 50
Magnolia soulangiana	18.2	4.4	3.9	6.2	90.2	25
Magnolia sp.	4	3.4	2.8	1.3	2.1	25
Malus pumila	19.6	5.3	4.4	4.4	177.3	25
Malus sp.	14.4	4.5	3.4	3.6	168.1	25

Table B.7 Continued

Species	DBH (cm)	Height (m)	Crown height (m)	Crown width (m)	Biomass (kg)	Biomass equation reference(s)
Morus alba	12.6	7.7	6.9	5.6	47.1	25
Morus rubra	35.6	10.7	9	9.6	477.6	25
Picea abies	53.2	15.3	14.6	10.3	959	4,32
Picea glauca	45.1	12.9	12.4	7.9	678.3	3,18,21,34,35,42
Picea glauca (dwarf)	9.1	2.6	2.6	1.6	16.7	3,18,21,34,35,42
Picea pungens	28	8.5	8.1	4.6	415.1	27
Picea sp.	11.3	5.8	5.7	3.5	28.8	27
Pinus contorta	18.5	9.7	8.7	3.4	60.5	7,19
Pinus nigra	20.6	8	6.6	3.9	187.6	7,19,20
Pinus ponderosa	45.7	15.6	12.4	6.9	645.7	7,19,20
Pinus sylvestris	30.8	9.6	8.1	7.5	227.1	4,12,38,41
Platanus occidentalis	41.2	12.9	10.9	9.9	436.8	15
Platycladus orientalis	17	2.9	2.8	1.2	46.6	33,42,44
Populus alba	34.9	17.5	18.2	6.6	557.6	23
Populus nigra	39	16.1	15	6.9	850.8	23
Populus sp.	23.8	13	11.9	4	171.1	23
Populus tremuloides	10.1	7.5	6	2.4	25.3	31
Prunus armeniaca	27.8	6.7	4.9	6.5	258.4	25
Prunus persica	6.3	4.1	3.4	2.1	9.5	25
Prunus sp.	12	5.1	3.9	3.5	63.3	25
Pseudotsuga menziesii	32.2	11.5	10.5	6.1	365.3	1,7,17,19,39
Pyrus calleryana	13	7.1	5.9	3.5	62.5	25
Pyrus communis	19.6	5.5	4.5	3.9	124.4	25
Quercus gambelii	10.2	4.9	3.1	2.5	44.6	11
Quercus macrocarpa	59.9	19.4	17	11.6	1317.2	42
Quercus palustris	2.5	2.4	1.4	1.6	1	24
Salix matsudana	38.4	9.5	8.7	9.8	768.7	23
Salix sepulcralis	132.6	18.8	18.8	9.9	10266.7	23
Salix sp.	124.8	11.6	11.6	9.8	5802.8	42,51
Sorbus aucuparia	14.5	6.4	5.1	3.9	71.7	25
Tamarisk sp.	10.6				24.2	29
Taxus baccata	2.6	2.7	2.7	1.3	2	29
Thuja plicata	3.5	1	0.9	1.5	1.8	7
Tilia cordata	5.2	3.7	2.1	1.9	4	25
Ulmus pumila	32.5	13.7	9.7	5.3	632.7	42,44
unknown hardwood	14.8	6.8	5.3	3.8	58.3	25
Zelkova serrata	26.7	10.1	8.7	9.9	233.8	25

Table B.8 References for biomass equations used in this study

Ref. no.	Author(s)	Ref. no.	Author(s)
1	Barclay et al. 1986	27	Jenkins et al. 2003, Spruce
2	Bartelink 1997	28	Jenkins et al 2003, True fir/hemlock
3	Baskerville 1965	29	Jenkins et al. 2003, Woodland
4	Brække 1986	30	Johansson 1999
5	Brenneman et al. 1978	31	Johnston and Bartos 1977
6	Bridge 1979	32	Jokela et al. 1986
7	Brown 1978	33	Ker 1980a
8	Canadell et al. 1988	34	Ker 1980b
9	Cerny 1990	35	Ker 1984
10	Chojnacky 1984	36	Kinerson and Bartholomew 1977
11	Chojnacky and Moisen 1993	37	MacLean and Wein 1976
12	Chroust 1985	38	Makela and Vanninen 1998
13	Cienciala et al. 2005	39	Marshall and Wang 1995
14	Clark et al. 1985	40	Martin et al. 1998
15	Clark et al. 1986a	41	Oleksyn 1999
16	Crow and Erdmann 1983	42	Perala and Alban 1994
17	Espinosa-Bancalari and Perry 1987	43	Phillips 1981
18	Freedman et al. 1982	44	Reiners 1972
19	Gholz et al. 1979	45	Reynolds et al. 1978
20	Grower et al. 1993	46	Ribe 1973
21	Harding and Grigal 1985	47	Schmitt and Grigal 1981
22	Hocker and Earley 1983	48	Wang et al. 1996
23	Jenkins et al. 2003, Aspen/alder/cottonwood/willow	49	Wiant et al. 1977
24	Jenkins et al. 2003, Hard maple/oak/hickory/beech	50	Williams and McClenahan 1984
25	Jenkins et al. 2003, Mixed hardwood	51	Young et al. 1980
26	Jenkins et al. 2003, Soft maple/birch		

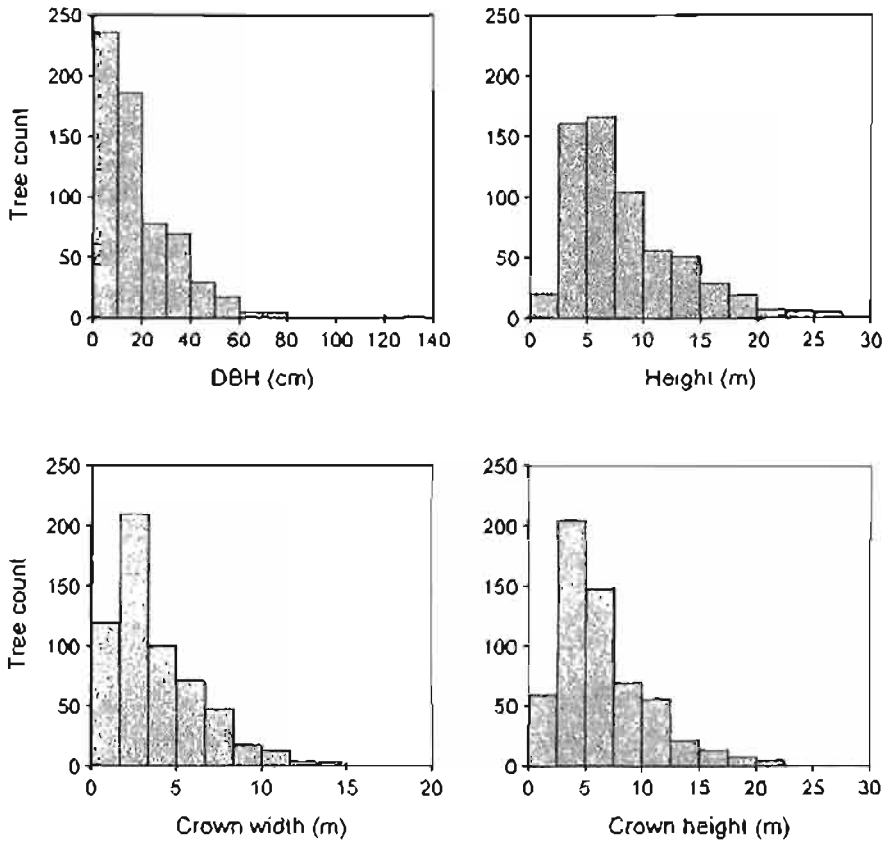

Fig. B.1 Study tree distribution by DBH, height, crown width, and crown height. Crown width is the average of two perpendicular crown measurements: one made N-S and the other E-W. Crown height is the distance from the lowest live foliage to the top of the tree.

References

Barclay H, Pang P, Pollard D (1986) Aboveground biomass distribution within trees and stands in thinned and fertilized Douglas fir. Canadian Journal of Forest Research 16:438-442

Bartelink HH (1997) Allometric relationship for biomass and leaf area of beech (Fagus sylvatica L). Annals of Forest Science 54:39-50

Baskerville G (1965) Dry-matter production in immature balsam fir stands. Forest Science Monographs 9

Brække FH (1986) Distribution and yield of biomass from young Pinus sylvestris and Picea abies stands on draines and fertilized peatland. Scandinavian Journal of Forest Research 1:49-66

Brenneman BB, Frederick DJ, Gardner WE, Schoenhofen LH, Marsh PL (1978) Biomass of species and stands of West Virginia hardwoods. In: Pope PE (ed) Proceedings of central hardwood forest conference II. Purdue University, West LaFayette pp 159-178

Bridge JA (1979) Fuelwood production of mixed hardwoods on mesic sites in Rhode Island. M.S. Thesis. University of Rhode Island, Kingston, Rhode Island, 72 pp

Brown J (1978) Weight and density of crowns of Rocky Mountain conifers. USDA Forest Service Resource Paper INT-197

Canadell J, Riba M, Andrés P (1988) Biomass equations for Quercus ilex L. in the Montseny Massif, Northeastern Spain. Forestry 61:137-147

Cerný M (1990) Biomass of Picea abies (L.) Karst. in midwestern Bohemia. Scandinavian Journal of Forest Research 5:83-95

Chojnacky D (1984) Volume and biomass for curlleaf cercocarpus in Nevada. USDA Forest Service Resource Paper INT-332

Chojnacky DC, Moisen GG (1993) Converting wood volume to biomass for pinyon and juniper. USDA Forest Service Research Note INT-411

Chroust L, Tesarova J (1985) Quatification of above-ground components of 20 years old Norway spruce (Picea abies (L.) Karsten). Communicationes Instituti Forestalis Cechosloveniae 14:111-126

Cienciala E, Cerný M, Alptauer J, Exnerová Z (2005) Biomass functions applicable to European beech. Journal of Forest Science 51: 147-154

Clark AI, Phillips D, Frederick D (1985) Weight, volume, and physical properties of major hardwood species in the Gulf and Atlantic Coastal Plains. USDA Forest Service Resource Paper SE-250

Clark AI, Phillips D, Frederick D (1986a) Weight, volume, and physical properties of major hardwood species in the Piedmont. USDA Forest Service Resource Paper SE-255

Crow TR, Erdmann GG (1983) Weight and volume equations and tables for red maple in the Lake States. USDA Forest Service Resource Paper NC-242:14

Espinosa Bancalari MA, Perry DA (1987) Distribution and increment of biomass in adjacent young Douglas-fir stands with different early growth rates. Canadian Journal of Forest Research 17:722-730

Freedman B, Duinker PN, Barclay H, Morash R, Prager U (1982) Forest Biomass and nutrient studies in central Nova Scotia. Canadian Forest Service Maritime Forest Research Center Inf Report M-X-134:126

Gholz HL, Grier CC, Campbell AG, Brown AT (1979) Equations for estimating biomass and leaf area of plants in the Pacific Northwest. Oregon State Univ. School of Forestry Forest Research Laboratory Research Paper No. 41

Gower ST, Reich PB, SON Y (1993) Canopy dynamics and aboveground production of five tree species with different leaf longevities. Tree Physiology 12:327-345

Harding RB, Grigal DF (1985) Individual tree biomass estimation equations for plantation-grown white spruce in northern Minnesota. Canadian Journal of Forest Research 15:738-739

Hocker HW, Early DJ (1978) Biomass and leaf area equations for northern forest species. University of New Hampshire, New Hampshire Agricultural Experimental Station Research Report Number 102

Jenkins JC, Chojnacky DC, Heath LS, Birdsey RA (2003) National -scale biomass estimators for United States tree species. Forest Science 49:12-35

Johansson T (1999) Biomass equations for determining functions of pendula and pubescent birches growing on abandoned farmland and some practical implicatons. Biomass and Bioenergy 16: 223-238

Johnston R, Bartos DL (1977) Summary of nutrient and biomass data from two aspen sites in the western United States. USDA Forest Service Research Paper INT-227

Jokela EJ, Van Gurp KP, Briggs RD, White EH (1986) Biomass estimation equations for Norway spruce in New York. Canadian Journal of Forest Research 16:413-415

Ker M (1980a) Tree biomass equations for ten major species in Cumberland County, Nova Scotia. Maritime Forest Research Center Inf Report M-X-108

Ker M (1980b) Tree biomass equations for seven species in southwestern New Brunswick. Maritime Forest Research Center Inf Report M-X-114

Ker M (1984) Biomass equations for seven major maritimes tree species. Canadian Forest Service Maritime Forest Research Center Inf Report, M-X-148:54

Kinerson A, Bartholomew I (1977) Biomass estimation equations and nutrient composition of white pine, white birch, red maple, and red oak in New Hampshire. University of New Hampshire, New Hampshire Agricultural Experimental Station Research Report Number 62

Maclean DA, Wein RW (1976) Biomass of jack pine and mixed hardwood stands in northeastern New Brunswick. Canadian Journal of Forest Research 6:441-447

Mäkelä A, Vanninen P (1998) Impacts of size and competition on tree form and distribution of aboveground biomass in Scots pine. Canadian Journal of Forest Research 28: 216-227

Marshall PL, Wang Y (1995) Above ground tree biomass of interior uneven-aged Douglas-fir stands. Canada-British Columbia Partnership Agreement on Forest Resource Development, University of British Columbia Working Paper WP-1.5-003

Martin J, Kloeppel B, Schaefer T, Kimbler D, McNulty S (1998) Aboveground biomass and nitrogen allocation of ten deciduous southern Appalachian tree species. Canadian Journal of Forest Research 28:1648-1659

Oleksyn J, Reich PB, Chalupka W, Tjoelker MG (1999) Differential above- and below-ground biomass accumulation of European Pinus sylvestris populations in a 12-year-old provenance experiment. Scandinavian Journal of Forest Research 14: 7-17

Perala DA, Alban DH (1994) Allometric biomass estimators for aspen-dominated ecosystems in the upper Great Lakes. USDA Forest Service Research Paper NC-314

Phillips D (1981) Predicted total–tree biomass of understory hardwoods. USDA Forest Service Research Paper SE-223

Reiners WA (1972) Structure and energetics of three Minnesota forests. Ecolological Monographs 42:71-94.

Ribe J (1973) Puckerbrush weight tables. University of Maine, Life Science and Agricultural Experimental Station Miscellaneous Report 152

Schmitt MDC, Grigal DF (1981) Generalized biomass estimation equations for Betula papyrifera Marsh. Canadian Journal of Forest Research 11:837-840

Wang JR, Zhong AL, Simard SW, Kimmins JP (1996) Aboveground biomass and nutrient accumulation in an age sequence of paper birch (Betula Papyrifera) in the Interior Hemlock zone, British Columbia. Forest Ecology and Management 83:27-38

Whittaker RH, Bormann FH, Likens GE, Siccama (1974) The Hubbard Brook ecosystem: Forest biomass and production. Ecological Monographs 44:223-252

Wiant HV, Sheetz CE, Colaninno A, DeMoss JC, Castaneda F (1977) Tables and procedures for estimating weights of some Appalachian hardwoods. West Virginia Agricultural Experimental Station Bulletin 659(T):36

Williams RA, McClenahen JR (1984) Biomass prediction equations for seedlings, sprouts, and saplings of ten central hardwood species. Forest Science 30:523-527

Young HE, Ribe JH, Wainwright K (1980) Weight tables for tree and shrub species in University of Maine, Life Science and Agricultural Experimental Station Miscellaneous Report 230